Report Rapport

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Commission de contrôle
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DERIVATION OF RELEASE LIMITS FOR A TYPICAL URANIUM MINING AND MILLING FACILITY

by

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ABSTRACT

DERIVATION OF RELEASE LIMITS FOR A TYPICAL URANIUM MINING AND MILLING FACILITY

This report develops guidelines for calculating derived release limits (DRLs) for releases of each radionuclide belonging to the uranium-238 and thorium-232 decay chains to atmosphere, surface water and groundwater from uranium mining and milling operations in Canada. DRLs are defined as calculated limits on releases from the facility that result in radiation exposures through all environmental pathways equal to the annual effective dose equivalent limit of 0.005 Sv for stochastic effects or the annual dose equivalent limit of 0.05 Sv for non-stochastic effects in the critical group.

By definition, DRLs apply to controllable radionuclide emissions which occur during the operational phase of mine/mill facilities. DRLs do not address the release of radionuclides to groundwater. Such releases are uncontrolled and may not have an impact on the critical group until long after close-out of the mine and mill. Uncontrolled releases were considered in this report by introducing the concept of an emission limit that will form one of the design constraints for tailings areas.

The concept of an emission limit for uncontrolled releases is distinct from but complementary to the derived release limit.

Radionuclides may be released to the atmosphere from underground and open-pit mining, from ore haulage activity, from milling operations, and from waste rock and tailings areas. Releases to surface water occur primarily through treated tailings and process water. Releases to groundwater occur largely by infiltration of atmospheric precipitation through waste rock and tailings areas, although mine water and drainage from the mill site may also be important seepage sources.

The report develops a steady-state environmental transfer model to determine environmental dilution and dispersion in atmosphere, surface water and groundwater between the sources at the mine and mill and the critical group receptor. Exposure pathways incorporated in the model include external exposure from immersion in the airborne plume, immersion in water, contaminated ground and contaminated shoreline sediments. Internal exposure pathways include inhalation of contaminated air and ingestion of contaminated water and terrestrial and aquatic foods.

Default values or modelling approaches are provided for each environmental transfer parameter for describing transport between environmental compartments. Included is a presentation of models to calculate dispersion in the atmosphere from point source and area source releases, and dispersion of aqueous releases in riverine, lake, estuarine and groundwater environments. Other parameters describe transfers between the other compartments listed above. Wherever possible, the user is urged to determine site-specific transfer parameter values rather than using the default values provided to ensure that the model more accurately reflects site-specific environmental conditions.

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RÉSUMÉ

Le présent rapport traite de lignes directrices établies pour calculer les limites de rejet dérivées (LRD) pour le rejet de chaque radionucléide de la chaîne de désintégration de l'uranium 238 et du thorium 232 dans l'air, les eaux superficielles et les eaux souterraines par suite de l'exploitation de mines et d'usines de concentration d'uranium au Canada. On peut définir les limites de rejet dérivées comme étant des calculs estimatifs des rejets qui proviennent d'une installation quelconque et qui provoquent des radio-expositions par toutes les voies d'acheminement environnementales, égales à la limite annuelle de l'équivalent de dose effectif de 0.005 Sv pour les effets stochastiques ou à la limite annuelle de l'équivalent de dose de 0,05 Sv pour les effets non stochastiques chez le groupe critique.

Par définition, les limites de rejet dérivées s'appliquent aux émissions de radionucléides maîtrisables qui se produisent durant la phase d'exploitation de la mine et de l'usine de concentration. Les limites de rejet ne s'appliquent pas aux rejets de radionucléides dans les eaux souterraines. Ces rejets ne sont pas contrôlés et peuvent n'avoir des répercussions sur le groupe critique que très longtemps après la fermeture de la mine et de l'usine de concentration. Le rapport tient compte des rejets non contrôlés en introduisant le concept d'une limite de rejet dans les contraintes de conception des aires de résidus.

Le concept d'une limite de rejet dans le cas des rejets non contrôlés n'est pas le même que celui de la limite de rejet dérivée, mais il le complète.

Les radionucléides peuvent être rejetés dans l'air à partir de mines souterraines ou à ciel ouvert, du roulage et du broyage du minerai, ainsi que des zones de stériles et de résidus. Les rejets vers les eaux superficielles se produisent surtout par l'intermédiaire des résidus traités et de l'eau de production. Les rejets vers le réseau des eaux souterraines surviennent les zones de stériles et de résidus, bien que l'eau de mine et l'eau de drainage du site de l'usine de concentration peuvent également devenir des sources importantes de fuite.

Le rapport met au point un modèle stable de transfert environnemental destiné à déterminer la dilution et la dispersion environnementales dans l'air, les eaux superficielles et les eaux souterraines entre les sources qui se trouvent à la mine et à l'usine de concentration et le groupe critique récepteur. Les voies d'acheminement intégrées au modèle comprennent l'irradiation externe due à l'immersion dans le panache en suspension dans l'air, l'immersion dans l'eau, ainsi que la contamination du sol et des sédiments sur le littoral. Les voies d'acheminement interne comprennent l'inhalation d'air contaminé et l'ingestion d'eau ou d'aliments terrestres ou aquatiques contaminés.

Le rapport fournit des valeurs implicites ou des approches de modélisation pour chaque paramètre de transfert environnemental pour décrire le transport entre les compartiments environnementaux. Il présente également des modèles pour calculer la dispersion dans l'atmosphère à partir d'un point et d'une zone de rejet, ainsi que la dispersion des rejets aqueux dans l'eau des rivières, des lacs, des estuaires et des réseaux d'eaux souterraines. D'autres paramètres définissent les transferts entre les autres compartiments mentionnés ci-dessus. On recommande fortement à l'utilisateur de déterminer, dans la mesure du possible, des valeurs paramétriques de transfert précises sur le site plutôt que d'utiliser les valeurs implicites pour s'assurer que le modèle traduit plus exactement les conditions environnementales précises du site.

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1.0 OBJECTIVES AND SCOPE

The CSA (Canadian Standards Association) Standard N288.1 sets forth guidelines for calculating Derived Release Limits (DRL's) for radioactive materials in airborne and liquid effluents for nuclear facilities, which include CANDU nuclear power plants, research reactors, radioisotope processing facilities and some types of waste processing facilities. However, the standard is not comprehensive enough to address uranium mining and milling.

The purpose of this document is to develop the methodology for calculating DRL's for uranium mines and mills. Although it complements CSA N288.1, it is not intended to be a standard. Its primary purpose is to serve as a technical basis document for use by Atomic Energy Control Board staff. In the context of this document, the DRL means the upper limit for the release rate of a single radionuclide from a single source which is derived from regulatory dose equivalent limits by analytical models of all significant environmental pathways to an individual in the most heavily exposed, or "critical", group.

The radionuclides considered in this document include all of the U-238 and Th-232 decay series. Radionuclides of the U-235 decay series are not explicitly included because U-235 constitutes only about 0.7%, on a mass basis, of the total uranium present in mines and mills. However, the DRL for U-NAT includes contribution from U-235 (see Section 4.3).

In addressing the releases of radionuclides from uranium mines and mills and their transport in the environment, several factors need to be considered that are different from those considered in CSA N288.1. Sources such as mine and mill stacks and vents and process or dewatering discharges may be classified as point sources, and can be addressed generically in a manner similar to that in CSA N288.1. However, other sources, such as area sources associated with tailings and waste rock areas, need special considerations. In addition to atmospheric and surface water dispersion and transport pathways, this document will also address groundwater transport.

In order to facilitate the use of this document, and in the interest of standardizing the approach used in calculating DRL's, the format of presentation of this document follows as closely as possible that used in CSA N288.1. Also, environmental transport models and default parameter values used in CSA N288.1, which are applicable to this document, have been adopted with little or no modification. This is considered reasonable and desirable since such models and parameter values had undergone extensive review during the preparation of N288.1. Default parameter values used in this report will be revised as new scientific data become available.

2.0 BASIC CONCEPTS

2.1 General

Derived Release Limits (DRL's) are calculated based upon the more restrictive of either the stochastic or non-stochastic dose limits using methodology described in Section 4.0. Because the DRL calculations require several individual steps, each of which often incorporates a measure of conservatism which is applied sequentially, the DRL is usually conservative. In assembling the reference data for this report, care was taken to ensure the values quoted were of a 'reasonable' nature. However, models and data of a site-specific nature should be used where these are available. In some cases, site studies may be warranted.

Although the approach to the calculation of DRL in this document is based on the calculated dose to a typical member of a critical group from a controllable routine release of radionuclides from an operating mine/mill facility, collective dose should be considered in the achievement of As Low As Reasonably Achievable (ALARA) objectives. This will involve the application of a process of optimization (ICRP, 1977), and is outside the scope of this document.

Care must also be taken to ensure that management options chosen to achieve the DRL or a designated fraction of the DRL during facility operation consider the long-term or post-abandonment radiological consequences of these options. For example, a mill may operate at 10% of its DRL with no barium-radium treatment, and may achieve 1% of its DRL with treatment. However, in the latter case, radium-enriched sludges will be produced that may present a long-term (post-abandonment) hazard. Thus, the benefit of short-term gains must be weighed against potential future risks.

2.2 Design Release Limits

It is in the area of concern of releases occurring over the long term, after the mining and milling operation has been abandoned, which necessitates a concept which transcends the DRL. Whereas the DRL relates to routine operational releases that are controllable through process modifications or more restrictive operating conditions, certain more long term releases are not covered by this concept. Such long-term, uncontrolled releases must be considered at the environmental impact assessment stage of a proposed uranium mine/mill project.

Perhaps the clearest example is the release of radionuclides from a tailings management area through groundwater seepage. In this case the rate of movement of radionuclides may be such that the radionuclides do not move beyond the tailings area until after abandonment, and any operational monitoring program would show no release from the site. Modelling predictions based upon data from other older sites would support that a plume of radionuclides will move out from below the tailings area , mix and dilute with local and regional groundwater, adsorb to geologic materials and decay. The sum of all of these processes will result in a forecast flux of radionuclides arriving at a critical group at some time in the future. Whether or not this forecast concentration results in a dose which would exceed the operational DRL depends on two basic factors: 1) the concentration of radionuclides at the source which is the engineered boundary of the tailings management system and 2) the nature of the geologic material and groundwater flow system which lies between the source and the critical group. This second aspect cannot be controlled to any significant degree. However, the concentration of radionuclides at the source can be controlled to some degree through mechanisms such as pH control of tailings pore water or the inclusion of a liner at the base of the tailings. The incorporation of these control options must be included as part of the facility design looking ahead to predicted post-closure effects.

Because future emissions are not monitored or controlled, the DRL concept does not strictly apply. Rather, the design criteria are based upon a forecast emission and the most likely effect on a critical group at some time in the future. The predicted effect on a future critical group and the design that would result in an acceptable effect could be referred to as a 'design release limit' or design emission limit.

In concept, the factors which dilute, retard (decay) and disperse radionuclides between the source (tailings) and the critical group would be considered. A dose of 5 mSv might be predicted to result from a release of X Bq/a of a particular radionuclide to the groundwater underlying a tailings management area. The facility design release limit would then be X Bq/a and a tailings management design would be selected that would result in some percentage (less than 100) of X Bq/a being released.

This concept of a design emission limit is thus applied to releases which are in effect uncontrollable after the facility is in operation. The effect of these releases on a critical group is not expected to occur until after abandonment. The design emission limit is presented here only as a concept and is not discussed in further detail in this report.

2.3 Dose Limits

The dose limits for members of the public are the basis for deriving release limits for effluent releases from mining and milling facilities. The limits, which have been recommended by the International Commission on Radiological Protection (ICRP, 1977) and have been included in a proposed amendment to the Atomic Energy Control Regulations (AECB, 1983), are specified in terms of annual individual doses for stochastic and non-stochastic effects as follows.

The "Stochastic Limit"

The effective dose equivalent received in a year from external sources of radiation, plus the committed effective dose equivalent from all prescribed substances that enter the body during the year shall not exceed 0.005 Sv.

The "Non-Stochastic Limit"

The dose equivalent received by any organ or tissue in a year from external sources of radiation, plus the committed dose equivalent to that organ or tissue from all prescribed substances that enter the body during the year shall not exceed 0.050 Sv.

2.4 The Critical Group Concept

The concept of the Critical Group is based upon the following statement provided by the ICRP (ICRP, 1977, paragraph 85):

> "With exposure of members of the public, it is usually feasible to take account of these sources of variability by the selection of appropriate critical groups within the population, provided the critical group is small enough to be relatively

> homogeneous with respect to age, diet and those aspects of behaviour that affect the doses received. Such a group should be representative of those individuals in the population expected to receive the highest dose equivalent, and the Commission believes that it will be reasonable to apply the appropriate dose-equivalent limit for individual members of the public to the weighted mean dose equivalent of this group. Because of the innate variability within an apparently homogeneous group, some members of the critical group will, in fact, receive dose equivalents somewhat higher than the mean. However, because of the maximizing assumptions used, the dose equivalent actually received will usually be lower than the estimated dose equivalent."

There are two main criteria to be satisfied in the selection of critical groups:

- the critical group should be representative of those expected to receive the highest doses; and
- the critical group should be relatively homogeneous with respect to those factors that affect doses received.

Site-specific diet and habit surveys should, in general, be made to identify and characterize the critical group. The ICRP has not defined "relatively homogeneous". The following methods for selecting the critical group from survey data are based on Hunt and Shepherd (1980) and Preston et al. (1974).

For internal exposures, a range of diet or habits (such as food consumption rate) corresponding to a ratio of not more than three between the maximum observed and the mean characterizing the critical group is considered sufficient to satisfy the homogeneity criterion. Based on a review of metabolic data, Preston et al. (1974) suggested that if the maximum to mean ratio of food consumption rates or habits within a group is less than three, then the expected metabolic variations will be predominant in determining the spread in doses. Hence, the group so selected will be "relatively homogeneous", as required in the ICRP definition of the critical group. The mean consumption value for the critical group so selected is then used to calculate the Derived Release Limits. The range of dietary and cultural habits that may be encountered in defining a critical group in the vicinity of Canadian uranium mining and milling facilities is very diverse; consequently, the default diet and habitat data presented in this report are not intended to be all inclusive, and at best can serve only as a guide.

For inhalation, the air intake rates do not vary widely among adults or among children of the same age group. Default values taken from ICRP-23 (ICRP, 1975) and given in Section 5.6 for the adult and the one-year old child may be used for the critical group.

For external exposures, the above homogeneity criterion may not apply. This is because behaviour may be more variable than metabolism within a population.

Recognizing the cost and difficulty of carrying out extensive site-specific surveys in some applications, this document provides default diet and habit data, generally conservatively based, which may be considered to characterize the critical group. Such data should be used with circumspection since, in certain situations, the possibility of under-estimating doses exists.

Explicit recommendations on defining the size of critical groups appear to be unavailable in the literature. In surveys of a Cumbrian coastal community in the United Kingdom, Hunt et al. (1982) made 177 observations on fish consumption from which only 9 observations were used to define the critical group, using the maximum:average equals 3 cutoff method described above. The critical group defined for the consumption of crustaceans consisted of only 15 people, although 92 were surveyed. Out of 726 people consuming commercial fish, only 20 were selected for the critical group. Thus, it would seem that a minimum critical group size of 10-20 is acceptable, although the number of people surveyed to arrive at the critical group should be considerably larger.

3.0 SOURCES OF RADIONUCLIDE RELEASES FROM URANIUM MINING AND MILLING FACILITIES

3.1 Description of Mining and Milling Facilities

Uranium mining facilities in Canada are generally classified as underground or open pit operations. Both types release radionuclides to the open environment and are source terms for derived release limit calculations. Open pit operations constitute aerial atmospheric sources of radon and particulate radionuclides. In underground mining, the majority of radon gas and particulates are released to the atmosphere through the ventilation exhaust vents. These vents are considered point sources for atmospheric releases.

In all areas of Canada that are presently mined for uranium, the ore bodies lie within the saturated groundwater zone. In order for mining to proceed, the pit or underground operation must be dewatered resulting in the production of pit or mine water. If this water is discharged directly, it may constitute a radionuclide source release. More often, however, pit sump water and mine water are recycled to the milling operation and do not constitute a discrete environmental source.

If a mine is located within a relatively permeable geologic formation, it may not be practical from a water balance or geotechnical standpoint to operate without first dewatering the mining area to reduce water pressures and inflow rates. This is commonly accomplished through the installation and operation of one or more interceptor dewatering wells or, in some instances, trenches. These wells or trenches intercept water moving in toward the groundwater depression caused by the excavation of the mine. The water collected by interceptor wells or trenches may or may not contain significant quantities of radionuclides, depending upon the amount of trace mineralization contained in the surrounding geologic materials.

Haulage costs of uranium ore can be a significant part of the overall cost of a uranium mining and milling operation. This fact, coupled with the desirability of as much water recycle as feasible to reduce overall radionuclide and trace element releases, usually results in the milling facility being placed as close as practical to the mine. If the mine and mill are a significant distance apart (> 2 km), then haulage by covered truck is usually employed for transport from the mine to the mill. Data on both particulate losses and gaseous losses from covered trucks suggest that the line source associated with haul roads will be negligible (IEC BEAK, 1984). Where uncovered trucks are used for ore haulage within the mine-mill complex, the particulate and gaseous losses would be considered as part of an area source.

Uranium mining and milling in Canada typically involves the mine production of ore containing a level of uranium above a certain cutoff grade. Other material that is removed to reach the ore is generally termed waste rock or overburden. This waste rock may contain trace amounts of uranium mineralization and, depending upon the waste management practices employed, may or may not constitute a source for DRL calculation. Within an open pit mining operation, radon gas release and the release of particulates associated with drilling, blasting and haulage operations all contribute to the pit source. Radon gas and particulates generated from comparable operations underground are, for the most part, vented through exhaust ducts that may or may not be equipped for particulate removal. Sources for DRL calculation from mining operations may thus be considered to potentially include:

- an area source from the pit for particulates and gaseous radon;
- one or more point sources from underground mining for particulates and radon gas;
- an area source from the waste rock pile;
- an area source for groundwater infiltration underlying the waste rock area; and
- one or more point sources for interceptor trench or well or pit sump or mine water if it is discharged directly.

After broken ore is removed from the mine, it is transported to a crushing operation where it is broken into smaller pieces. Crushing may result in the release of radon gas and particulates. Crushing is followed by a grinding operation where the crushed ore is ground to a relatively fine particle size designed to optimize the efficiency of the uranium extraction. (Crushing and grinding operations are typically wet or semi-wet in nature to minimize dust or particulate loss. Wet crushing and grinding also aids in reducing loss of radon gas.) After the ore is ground, it passes through a leaching circuit. The leaching circuit is usually customized to suit the particular ore, but typically involves exposing the ore to high temperature sulphuric acid. Elevated pressures may also be used in combination with the strongly oxidizing environment. Following leaching is a liquid-solids separation step and recovery of the solute. The solids are typically transported to a tailings area, while the solute or 'pregnant' solution undergoes ion exchange for uranium recovery. The barren solution from which the uranium has been extracted is then recycled to the liquid-solid separation phase or disposed of in the tailings management area. Because this process is wet, no significant particulate releases are experienced, and radon gas is the only radionuclide of significance released. The eluate from the ion exchange circuit then undergoes a precipitation or stripping step in which the uranium is precipitated as either an ammonium diuranate or a magnesium diuranate. The stripped eluate is then recycled or diverted to the tailings area. This precipitated product is then dried and packaged as yellowcake. The drying and packaging cycle is the final point of radionuclide release to the atmosphere. This cycle is shown schematically in Figure 3.1.

In summary, the following milling processes represent sources of radionuclides to be included in DRL calculations:

- o crushing and grinding,
- o leaching, and
- o yellowcake drying and packaging.

3.2 Radioactive Waste Management Practice

Over the last couple of decades, waste management practices in Canadian uranium mines and mills have moved toward a maximization of water and aqueous stream recycle and a reduction of point source liquid discharges. The result has been that mine interceptor wells and/or trenches have reduced, in many cases, the amount of mine water that must be treated. Mine or pit sump water which cannot meet discharge standards often contains significant dissolved and suspended solids, and is rich in uranium, with lesser quantities of other dissolved radionuclides. This water is often used as mill make-up water. This results in additional product recovery in addition to an indirect cycling of the water through the tailings treatment system. The result is that all process streams are treated through one point of control, either as a tailings decant or as a treated process stream that has been recovered from tailings and other areas of the mill. In

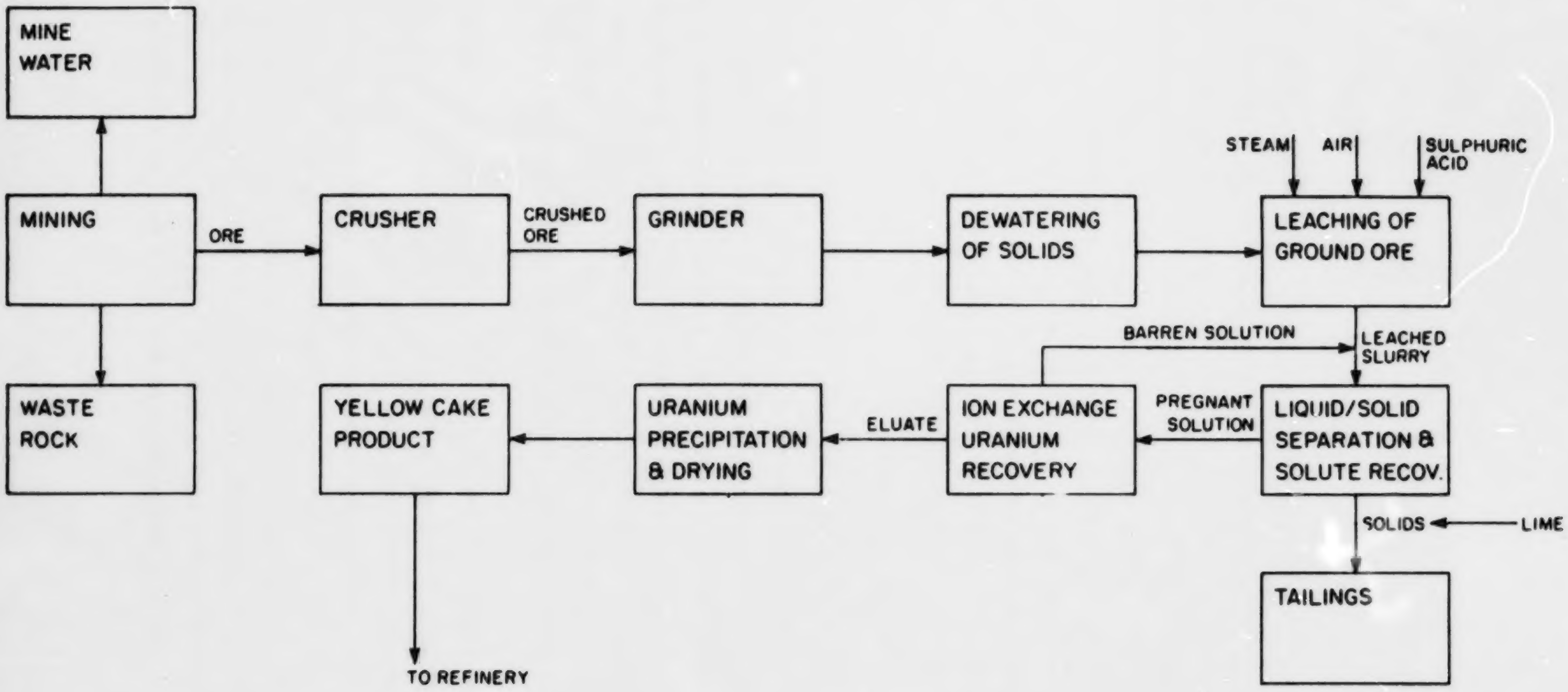
MINE WATER
MINING
WASTE ROCK
ORE
CRUSHER
CRUSHED ORE
GRINDER
DEWATERING OF SOLIDS
STEAM
AIR
SULPHURIC ACID
LEACHING OF GROUND ORE
BARREN SOLUTION
LEACHED SLURRY
LIQUID/SOLID SEPARATION & SOLUTE RECOV.
PREGNANT SOLUTION
ION EXCHANGE URANIUM RECOVERY
ELUATE
URANIUM PRECIPITATION & DRYING
YELLOW CAKE PRODUCT
TO REFINERY
SOLIDS
LIME
TAILINGS

FIGURE 3.1 Typical Acid Leach Milling Circuit

most cases, it is this final single point of release that is considered for aquatic point source DRL calculations. In some instances, interceptor well point source discharges may be considered.

In addition to the aqueous point of discharge, there may be several area or seepage sources. These typically underly the tailings area, waste rock area and other miscellaneous areas, such as holding ponds, the mill terrace or any other place where liquids containing radionuclides may seep into the ground. To control releases from these sources, liners of either natural materials (clays) or synthetics are used to restrict infiltration of water into the underlying ground. Operational areas, such as the mill terrace, may be paved to restrict infiltration from small local spills. The result is that the only significant losses to groundwater are usually found beneath the tailings and waste rock areas, with the magnitude of these losses dependent upon the type and integrity of liner used and the operational practices followed (e.g., hydraulic head present).

On the atmospheric side, release of radon gas from underground and open pit facilities is largely a function of the radium content of exposed rock, the emanation coefficient, atmospheric conditions, and the resultant radon exhalation rate. Very little option exists in the area of technology to control radon releases to the atmosphere from the mining operations. For worker health reasons, radon and its particulate daughters are vented to the atmosphere as fast as is practical. Fugitive dust from blasting, mucking and haulage operations contain particulate radionuclides. Loss of particulates is managed at present by dust suppressants and the control of speeds at which haulage trucks travel.

Crushing and grinding operations are carried out wet in order to suppress dust. Radon from crushing and grinding is released directly to the atmosphere. The leaching, solid liquid separation and elution processes are all wet processes, and fugitive dust losses from them are minimal. Radon released during these processes escapes directly to the atmosphere. The final step of yellowcake drying and packaging is relatively free of radon release, as the uranium product contains very little radium. Yellowcake dust releases are controlled through conventional bag house dust control technology. Loss of fugitive dust from the mill terrace area is commonly controlled in new facilities through paving of the mill terrace and revegetation of dewatered tailings, abandoned roadways and waste areas.

Tailings are, as a matter of practice, discharged wet and with proper management can be kept close to a full state of saturation. Saturated tailings will release very little radon gas because of radon's lower diffusion coefficient in water than in air. The high moisture content of tailings minimizes fugitive dust losses. The management practice of processing a low grade ore during the last year or so of mill operation may lead to production of a less radioactive tailings which may be used to cover the tailings mass. Thus, a less hazardous layer will be in direct contact with the environment. Other practices specific to certain ore types such as the surface liming of pyritic tailings or the application of surface covers may be carried out to reduce releases to the post-abandonment environment.

3.3 Radionuclide Releases to the Atmosphere

In calculating derived release limits for particular radionuclides, it is necessary to consider several sources and the nature of those sources. The sources may be considered to be in several distinct operations areas, including:

- mining,
- haulage,
- milling, and
- tailings waste management.

Mining activity may result in significant releases of radon gas and dust. If the mining is carried out underground, radon gas and any associated particulate releases will be from point sources. How these sources are modelled for DRL calculation purposes depends upon the location of the vents and their effective height. If the mining is carried out as an open pit operation, both the dust and radon will be released from a diffuse area source.

Ore from the mine is generally hauled to the mill by covered truck. Where hauling distances are less than 2 km, the haulage road can be considered as contributing to a general mine/mill area source. This is generally the case at most Canadian facilities. If haulage distances are longer and the ore trucks are uncovered or the ore is of high grade, radionuclide losses may be significant and a line source model may be required to assess atmospheric dispersion.

The milling operation, including crushing, grinding, extraction, recovery, drying and packaging, is carried out in most instances within a central area where sources are no more than several hundred metres apart, and releases occur from vents on the sides or tops of buildings. These sources for DRL calculation purposes may be considered discrete point sources as each of these sources can be separately monitored and controlled.

The waste management area may consist of a conventional above ground tailings facility with dry beach areas from which radon gas and particulate dusts may be released. These beach areas may be treated as area sources for which it is important to define several process and environmental variables such as grain size, specific activity, small grain particle size activity enhancement, surface roughness, moisture content, etc. Areas of tailings that are permanently submerged, as may exist in backfilled tailings in mined-out slopes or tailings placed under water in a lake basin or marine environment, need not be considered for DRL purposes as contributing particulates or radon gas. Only in special instances where tailings placed within a lake or marine environment result in a high level of suspended solids, and those solids are deposited on shorelines where they are subject to wind erosion or radon exhalation do these sources require consideration.

Recommendations of how these sources are to be treated for the calculation of derived release limits are presented in Appendix E.

3.4 Radionuclide Releases to Surface Waters

Modern mine and mill complexes in Canada, for the most part, have one or possibly two sources of surface water discharge. The first would be a treated water discharge which is comprised of treated tailings and process water. This water is usually closely monitored and regulated with respect to flow. The second source is interceptor water comprised of groundwater flowing toward the groundwater sink created by the open pit or underground mine. This water is generally considered to have "background" water quality as it has not yet come in contact with any aspect of the mining or milling operation. This source is considered for DRL calculations if it fails to meet surface water quality objectives for radionuclides, or if any significant elevation exists above background radionuclide levels.

3.5 Radionuclide Releases to Groundwater

Within the mine/mill complex, there are several areas where loss of radionuclides to groundwater may occur. The area of the mine itself during operations is generally a groundwater sink with groundwater flowing toward it and being pumped out by interceptor wells, or as pit or mine water. The only case where there may be significant losses to groundwater during mining would be solution mining, where recovery of lixiviant was low.

Little opportunity for significant loss of radionuclides to groundwater occurs during haulage operations, in the crushing and grinding or in the milling and product packaging. Spills may occur in the mill terrace area and, if it is not paved, some seepage loss may occur to groundwater. Only in special circumstances need this source be considered as a groundwater source for DRL calculation.

The waste rock and tailings areas are generally the two sources with the greatest potential for operating loss of radionuclides to groundwater. The high porosity and permeability of typical waste rock suggests that for DRL calculation purposes, all precipitation falling on the waste rock area must be considered to enter the geologic material underlying the waste rock unless there are data to the contrary.

Tailings management areas, in contrast to waste rock areas, often have controlled water budgets. During construction, the tailings area may be lined with a low permeability material. The tailings may also have an underdrain to intercept seepage waters. Furthermore, the tailings may be layered to reduce vertical permeability and, during operation, the hydraulic head can be managed to regulate seepage. The result is that the amount of seepage from tailings areas can vary significantly between mills. All of these factors must be considered when evaluating a source release for DRL calculations.

4.0 OVERVIEW OF METHODOLOGY

4.1 Generalized Environmental Transfer Pathway Model

Radioactive materials released to the environment are dispersed and transported through various environmental pathways. For the purpose of DRL calculations, only those pathways that would lead to radiation exposure to humans are of interest. These exposure pathways are:

1. external exposure from:
   - immersion in the airborne plume,
   - immersion in water,
   - contaminated ground,
   - contaminated shoreline;
2. inhalation; and
3. ingestion of terrestrial and aquatic foods.

Following the approach used in the CSA N288.1 Standard, a generalized radioactive material transport model has been developed for uranium mines and mills, shown in Figure 4.1. This approach facilitates systematic development of the environmental transport models, data presentation and future updates.

The quantity in each block or compartment i in Figure 4.1 is denoted by X_i. Definitions of X_i and units used are summarized in Table 4.1. Transfer from compartment i to compartment j is characterized by a transfer parameter P_{ij} such that the amount in compartment j arising from transfer from compartment i under steady-state conditions is $P_{ij} X_i$. Transfer parameters and their units are defined in Table 4.2. Thus, the quantity represented by any compartment j is given by:

$$X_j = \sum_i P_{ij} X_i \qquad (4.1)$$

where the summation is over all compartments transferring into compartment j. If all the P_{ij}'s are known, the quantity in any compartment for a given release rate X_S may be calculated. For example, in the case of releases to the atmosphere:

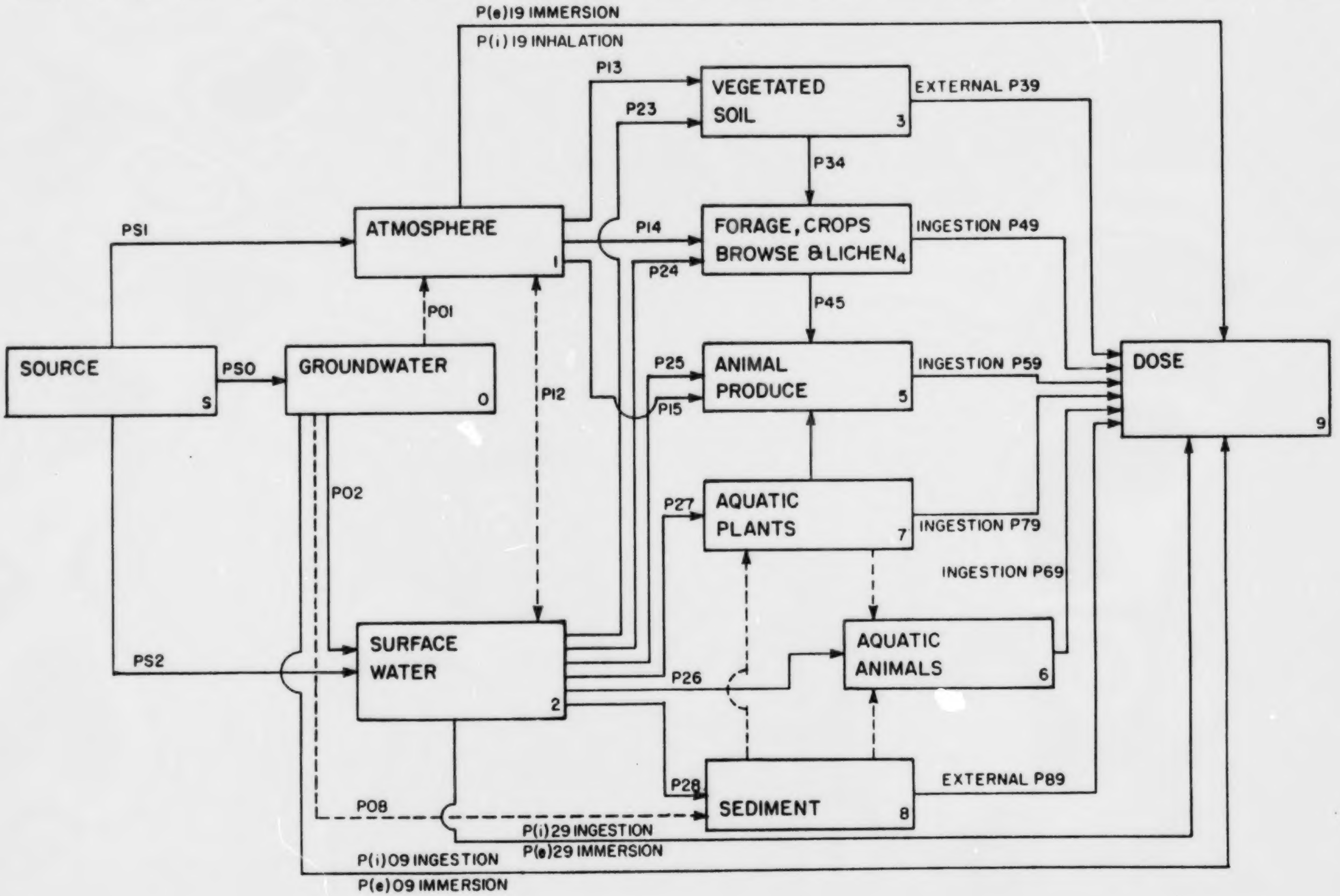
P(e)19 IMMERSION
P(i) 19 INHALATION
P13
P23
VEGETATED SOIL 3
EXTERNAL P39
P34
ATMOSPHERE 1
PS1
P14
P24
FORAGE, CROPS BROWSE & LICHEN 4
INGESTION P49
P45
P01
SOURCE S
PS0
GROUNDWATER 0
P12
P25
P15
ANIMAL PRODUCE 5
INGESTION P59
DOSE 9
P02
P27
AQUATIC PLANTS 7
INGESTION P79
INGESTION P69
SURFACE WATER 2
PS2
P26
AQUATIC ANIMALS 6
P28
SEDIMENT 8
EXTERNAL P89
P08
P(i)29 INGESTION
P(e)29 IMMERSION
P(i)09 INGESTION
P(e)09 IMMERSION

FIGURE 4.1 Generalized Environmental Transfer Pathway Diagram

TABLE 4.1: UNITS FOR GENERALIZED TRANSFER COMPARTMENTS

Compartment		Units
S	(Source)	$Bq \cdot s^{-1}$
0	(Groundwater)	$Bq \cdot L^{-1}$
1	(Atmosphere)	$Bq \cdot m^{-3}$
2	(Surface Water)	$Bq \cdot L^{-1}$
3	(Vegetated Soil)	$Bq \cdot m^{-2}$
4	(Forage, Crops, Browse and Lichen)	$Bq \cdot kg^{-1}$
5	(Animal Produce)	$Bq \cdot kg^{-1}$
6	(Aquatic Animals)	$Bq \cdot kg^{-1}$
7	(Aquatic Plants)	$Bq \cdot kg^{-1}$
8	(Sediment)	$Bq \cdot kg^{-1}$
9	(Dose)	$Sv \cdot a^{-1}$

Note: Replace Bq with ug in the case of U-NAT (see Section 4.3).

TABLE 4.2: UNITS FOR TRANSFER PARAMETERS

Transfer Parameter	Compartments	Units
P_{S1}	Source to Atmosphere	$s \cdot m^{-3}$
$P(i)_{19}$	Atmosphere to Dose (inhalation)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3$
$P(e)_{19}$	Atmosphere to Dose (immersion)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3$
P_{13}	Atmosphere to Vegetated Soil	m
P_{14}	Atmosphere to Forage, Crops, Browse + Lichen	$m^3 \cdot kg^{-1}$
P_{15}	Atmosphere to Animal Produce	$m^3 \cdot kg^{-1}$
P_{34}	Vegetated Soil to Forage, Crops + Browse	$m^2 \cdot kg^{-1}$
P_{39}	Vegetated Soil to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^2$
P_{45}	Forage, Crops, Browse + Lichen to Animal Produce	$kg \cdot kg^{-1}$
P_{49}	Forage + Crops to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$
P_{59}	Animal Produce to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$
P_{S2}	Source to Surface Water	$s \cdot L^{-1}$
P_{23}	Surface Water to Vegetated Soil	$L \cdot m^{-2}$
P_{24}	Surface Water to Crops	$L \cdot kg^{-1}$
P_{25}	Surface Water to Animal Produce	$L \cdot kg^{-1}$
P_{26}	Surface Water to Aquatic Animals	$L \cdot kg^{-1}$
P_{27}	Surface Water to Aquatic Plants	$L \cdot kg^{-1}$
P_{28}	Surface Water to Sediment	$L \cdot kg^{-1}$
$P(i)_{29}$	Surface Water to Dose (Ingestion)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$
$P(e)_{29}$	Surface Water to Dose (Immersion)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$
P_{69}	Aquatic Animal to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$
P_{79}	Aquatic Plant to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$
P_{89}	Sediment to Dose	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$
P_{S0}	Source to Groundwater	$s \cdot L^{-1}$
P_{02}	Groundwater to Surface Water	(unitless)
P_{03}	Groundwater to Vegetated Soil	$L \cdot m^{-2}$

TABLE 4.2: UNITS FOR TRANSFER PARAMETERS

Transfer Parameter	Compartments	Units
P_{04}	Groundwater to Crops	$L \cdot kg^{-1}$
P_{05}	Groundwater to Animal Produce	$L \cdot kg^{-1}$
$P(i)_{09}$	Groundwater to Dose (Ingestion)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$
P'_{09}	Groundwater to Dose (Inhalation)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$
$P(e)_{09}$	Groundwater to Dose (Immersion)	$Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$

Note: Replace Bq with ug in the case of U-NAT (see Section 4.3).

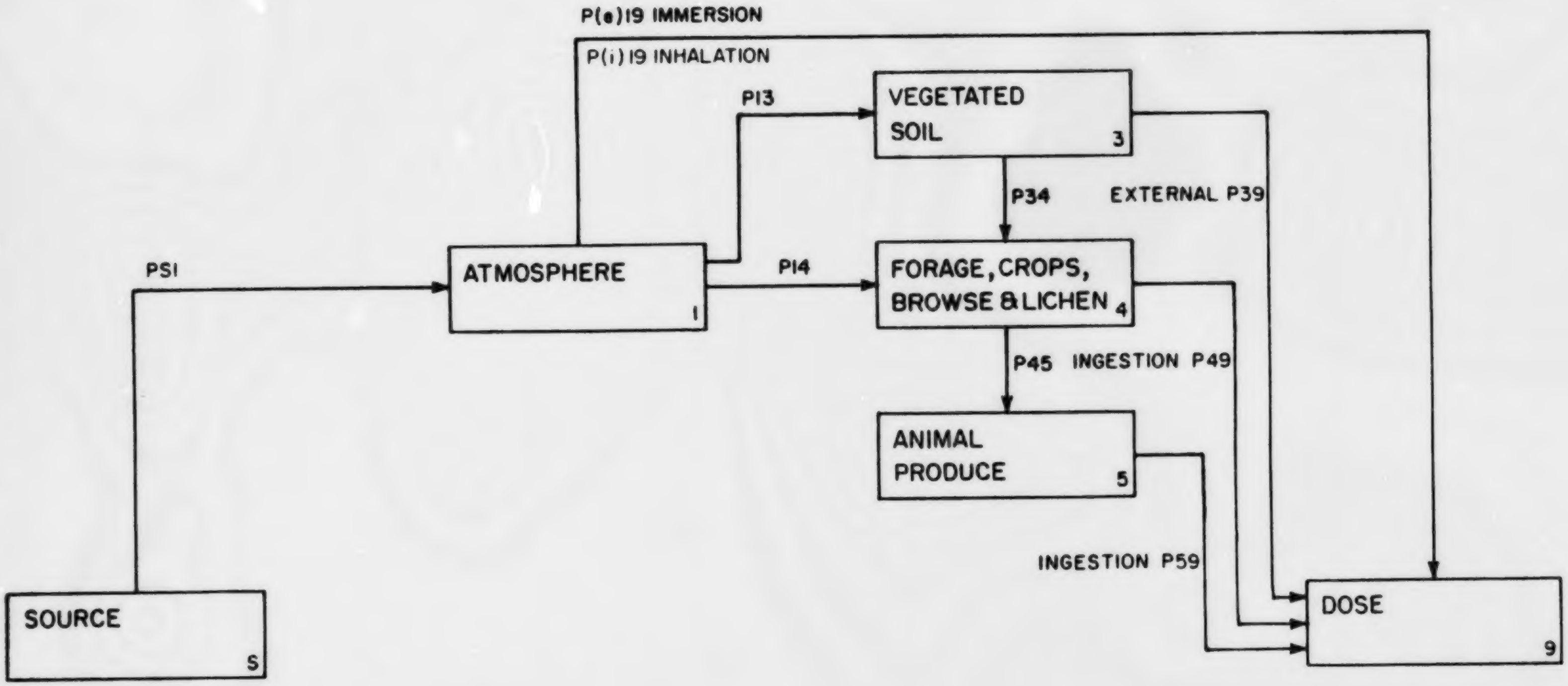
P(e)19 IMMERSION
P(i)19 INHALATION
PSI
P13
P14
VEGETATED SOIL 3
P34
EXTERNAL P39
ATMOSPHERE 1
FORAGE, CROPS, BROWSE & LICHEN 4
P45
INGESTION P49
ANIMAL PRODUCE 5
INGESTION P59
SOURCE S
DOSE 9

FIGURE 4.2 Simplified Environmental Transfer Pathways for Radionuclide Releases to the Atmosphere

$$\begin{aligned}
X_1 &= P_{S1}\, X_S(a) \\
X_3 &= P_{13}\, X_1 = P_{S1}\, P_{13}\, X_S(a) \\
X_4 &= P_{14}\, X_1 + P_{34}\, X_3 \\
&= P_{S1}\, P_{14}\, X_S(a) + P_{13}\, P_{34}\, X_1 \\
&= P_{S1}\,(P_{14} + P_{13}\, P_{34})\, X_S(a)
\end{aligned}$$

where $X_S(a)$ is the release rate to atmosphere. Similar expressions may be derived for releases to surface water $X_S(w)$ and to groundwater $X_S(g)$.

Transfer parameters indicated by dotted lines are conceptual pathways that are not explicitly calculated (Figure 4.1). Quantities of radionuclides transported between atmosphere and surface water or groundwater are considered insignificant in comparison with the quantities discharged directly to these compartments at the mine and mill. Transfers among the aquatic plants, aquatic animals and sediment compartments may be significant; however, concentrations in aquatic foods are calculated based on the ratio of the concentration in tissue to the concentration in surface water (bioconcentration factor). **In using this simple bioaccumulation model, it should be recognized that tissue contaminant burdens result from bioaccumulation through the food chain and from sediment as well as directly from water.**

The environmental transport models assume that steady-state conditions apply; however, in reality, non-steady state conditions may be expected to frequently occur. Nonetheless, for operational periods of years and decades, it is reasonable to assume that steady-state processes describe average conditions.

4.2 Calculation of Derived Release Limits (DRL)

The DRL is obtained by:

- applying the overall transfer model to the pathways loading from the source to humans;
- calculating the dose rate per unit release rate; and
- calculating the DRL.

The calculation of DRL is given herein for atmospheric, surface water and groundwater

The calculation of DRL is given herein for atmospheric, surface water and groundwater modes of transmission.

4.2.1 Calculation of DRL for Simplified Sets of Pathways

In this subsection, the generalized environmental transfer pathway diagram (Figure 4.1) is simplified to illustrate application of generalized environmental transfer models in the calculation of DRL. The general methodology for all pathways is given in Section 4.2.3.

It should be noted that the simplification given herein may not be appropriate for all environments. Each user must consider all pathways and prove that simplifications are in order, before selecting specific transfer pathways for the DRL calculations.

Release to Atmosphere

An environmental transfer diagram for the release of radionuclides to atmospheric pathways is given in Figure 4.2. The pathways used here in calculating the DRL for releases to the atmosphere ignore pathways leading to atmosphere from groundwater and surface water, since these sources are considered insignificant. For this example, the assumptions are also made that the pathway from atmosphere to animal produce to man is insignificant and that deposition to lakes and rivers and the associated dose to humans is negligible. Again, it should be emphasized that in any real application, the user must demonstrate that all pathways used in DRL calculations are appropriate for the specific, prevailing environmental conditions.

Application of the environmental transfer model (Equation 4.1) to these pathways (Figure 4.2) gives the dose to humans $X_9(a)$ through atmospheric pathways as:

$$X_9(a) \quad = \quad P(e)_{19} X_1 + P(i)_{19} X_1 + P_{39} X_3 + P_{49} X_4 + P_{59} X_5 \qquad (4.2a)$$

from compartments 1, 3, 4 and 5, respectively. This expression may be expanded as follows:

$$
\begin{aligned}
X_9(a) &= (P(e)_{19} + P(i)_{19})\, X_1 + P_{39}\, X_3 + (P_{45}\, P_{59} + P_{49})\, X_4 && (4.2b)\\
&= (P(e)_{19} + P(i)_{19})\, X_1 + (P_{39}\, P_{13}\, X_1) + (P_{45}\, P_{59} + P_{49})\\
&\quad P_{14}\, X_1 + (P_{45}\, P_{59} + P_{49})\, P_{34}\, X_3 && (4.2c)\\
&= (P(e)_{19} + P(i)_{19})\, X_1 + (P_{39}\, P_{13})\, X_1 + (P_{45}\, P_{59} + P_{49})\, P_{14}\, X_1\\
&\quad + (P_{45}\, P_{59} + P_{49})\, (P_{34}\, P_{13})\, X_1 && (4.2d)\\
&= (P(e)_{19} + P(i)_{19} + P_{39}\, P_{13} + P_{14}\, P_{45}\, P_{59} + P_{14}\, P_{49} +\\
&\quad P_{13}\, P_{34}\, P_{45}\, P_{59} + P_{13}\, P_{34}\, P_{49})\, X_1 && (4.2e)\\
&= (P(e)_{19} + P(i)_{19} + P_{39}\, P_{13} + P_{14}\, P_{45}\, P_{59} + P_{14}\, P_{49} +\\
&\quad P_{13}\, P_{34}\, P_{45}\, P_{59} + P_{13}\, P_{34}\, P_{49})\, P_{S1}\, X_S && (4.2f)
\end{aligned}
$$

The dose to humans per unit discharge through the atmospheric pathway is:

$$X_9(a)/X_S \qquad (4.3)$$

Hence, the DRL for atmospheric releases is:

$$\text{Atmospheric DRL} = \frac{\text{Annual Dose Limit}}{X_9(a)/X_S} \quad \text{Bq}\cdot\text{s}^{-1} \qquad (4.4)$$

Release to Surface Waters

A simplified environmental transfer diagram for the release of radionuclides to surface water pathways is given in Figure 4.3. The pathway shown here ignores the surface water to vegetated soil and forage and crops pathway (i.e., no irrigation) and the aquatic plants to animal produce pathway (e.g., moose used as a human food source). Transfer through the aquatic food chain from sediments to aquatic plants (e.g., wild rice) and aquatic animals (e.g., fish) are not explicitly included in this pathway; transfer parameters P_{26} and P_{27} are assumed to incorporate all routes for bioaccumulation of radionuclides in compartments 6 and 7.

Application of the environmental transfer model to these pathways (Figure 4.3), as demonstrated in detail for the calculation of $X_9(a)$ (Equation 4.2), gives the dose to humans for discharges to surface waters $X_9(w)$ as:

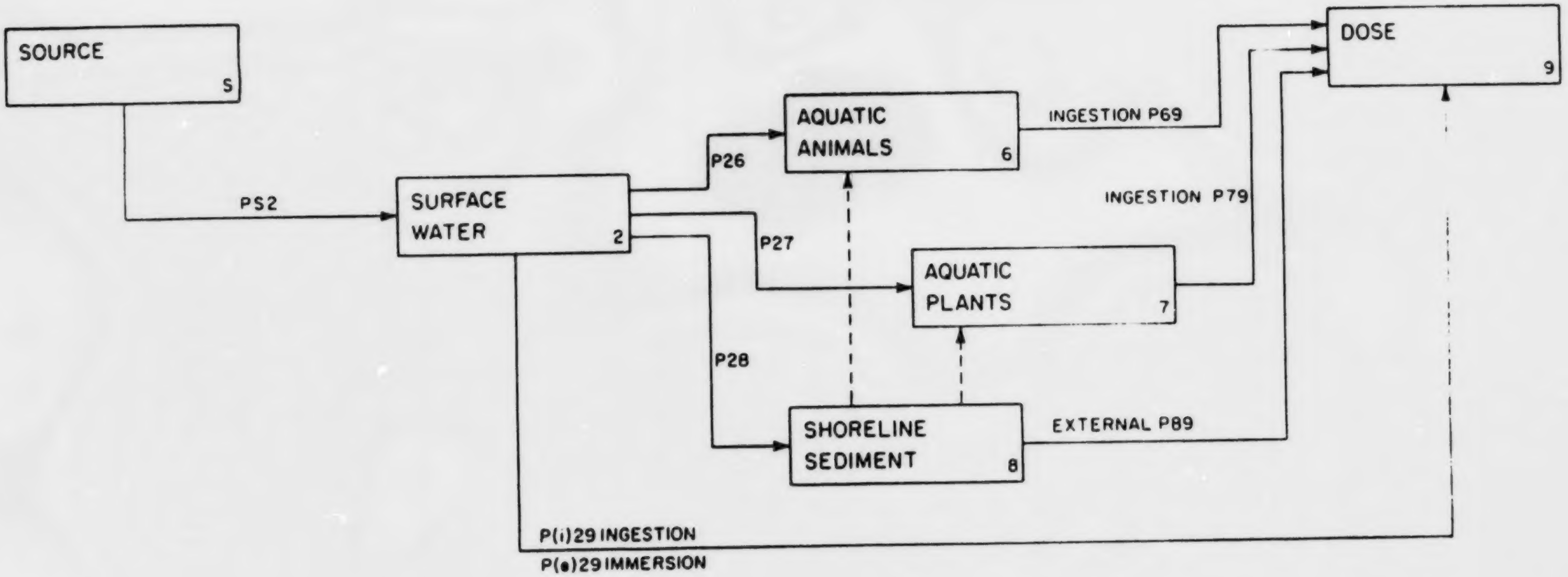
SOURCE
S
PS2
SURFACE WATER
2
P26
P27
P28
AQUATIC ANIMALS
6
AQUATIC PLANTS
7
SHORELINE SEDIMENT
8
INGESTION P69
INGESTION P79
EXTERNAL P89
DOSE
9
P(i)29 INGESTION
P(e)29 IMMERSION

FIGURE 4.3

Simplified Environmental Transfer Pathways for Radionuclide Release to Surfacewaters

$$X_9(w) = (P(e)_{29} + P(i)_{29})\, X_2 + P_{89}\, X_8 + P_{79}\, X_7 + P_{69}\, X_6 \tag{4.5a}$$

$$= (P(e)_{29} + P(i)_{29} + P_{28}\, P_{89} + P_{27}\, P_{79} + P_{26}\, P_{69})\; P_{S2} X_S \tag{4.5b}$$

The dose to humans per unit discharge to surface water is:

$$X_9(w)/X_S \tag{4.6}$$

The DRL for surface water releases is:

$$\text{Surface Water DRL} = \frac{\text{Annual Dose Limit}}{X_9(w)/X_S}\ \text{Bq·s}^{-1} \tag{4.7}$$

Release to Groundwaters

A simplified pathway diagram for radionuclide release to groundwater is given in Figure 4.4. Figure 4.4 assumes that the main human exposure results from well water delivered to a house, irrigation of crops and watering of livestock using well water as the source. The transfer to a watershed (compartments 3, 4, 5) from groundwater (compartment 0) is shown occurring through surface water (compartment 2). In fact, transfer may occur through surface water (e.g., an irrigation pond), or it may be direct (i.e., P_{03}, P_{04}, P_{05}). Radionuclide release from groundwater to atmosphere is considered to be negligible and is thus ignored in this pathway.

Another plausible set of pathways to humans is the discharge of a groundwater aquifer to a lake which then transfers the radionuclide to aquatic animals, plants and shoreline sediments (respectively, compartments 6, 7 and 8; Figure 4.1). Discharge to a lake would require a different value for P_{02} than that for groundwater to surface water via irrigation. These pathways are not considered herein, but may require consideration for particular environments.

Application of the environmental transfer model to these pathways (Figure 4.4) gives the dose to humans $X_9(g)$ for discharges to groundwater as:

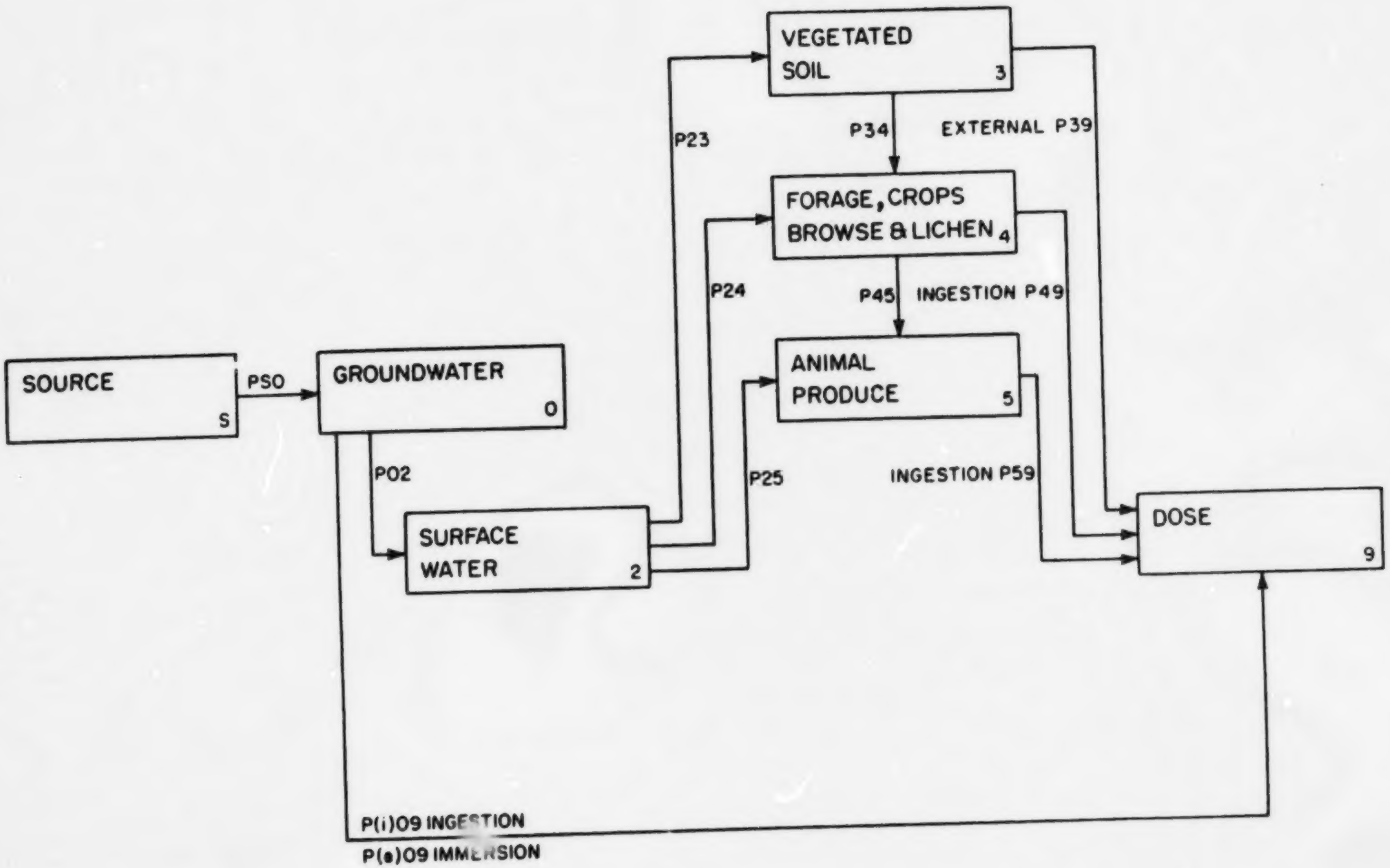
SOURCE
S
PSO
GROUNDWATER
0
P02
SURFACE WATER
2
P23
P24
P25
VEGETATED SOIL
3
P34
EXTERNAL P39
FORAGE, CROPS BROWSE & LICHEN
4
P45
INGESTION P49
ANIMAL PRODUCE
5
INGESTION P59
DOSE
9
P(i)09 INGESTION
P(e)09 IMMERSION

FIGURE 4.4 Simplified Environmental Transfer Pathway for Radionuclide Release to Groundwater

$$X_9(g) = P_{39} X_3 + P_{49} X_4 + P_{59} X_5 + P(e)_{09} X_0 + P(i)_{09} X_0 \quad (4.8a)$$

$$= ((P_{39} P_{23} + (P_{49} + P_{59} P_{45}) (P_{34} P_{23} + P_{24}) + P_{59} P_{25}) P_{02} + P(e)_{09} + P(i)_{09}) P_{S0} X_S \quad (4.8b)$$

The dose to humans per unit discharge to groundwaters is:

$$X_9(g)/X_S \quad (4.9)$$

The DRL for groundwater releases is:

$$\text{Groundwater DRL} = \frac{\text{Annual Dose Limit}}{X_9(g)/X_S} \ \text{Bq}\cdot\text{s}^{-1} \quad (4.10)$$

4.2.2 Stochastic and Non-Stochastic Effects for DRL Calculations

As discussed in Section 2.2, a distinction is made between stochastic effects, for which the Annual Effective Dose Equivalent Limit is 0.005 Sv, and non-stochastic effects, for which the Annual Dose Equivalent Limit is 0.05 Sv. For many radionuclides, the DRL will be limited by the stochastic dose limit. However, there are radionuclides for which the DRL will be limited by the non-stochastic limit for the dose to a particular organ. These radionuclides and the respective organs of concern are listed in Table B2 (Appendix B). For these radionuclides, separate DRL calculations should be made for stochastic effects and non-stochastic effects using the relevant Annual Dose Limits. The smaller of the two calculated values should then be used as the DRL.

The DRL calculation for stochastic effects in Equations 4.4, 4.7 or 4.10 will involve the effective dose conversion factors for internal exposures given in Table B1 (Appendix B) and the appropriate effective dose conversion factors for external exposures given in Tables B3, B4 or B5. The DRL calculation for non-stochastic effects will involve the organ dose conversion factors for internal exposures given in Table B2 (Appendix B) and the selection of dose conversion factors for doses to the same organs due to external exposures. The appropriate effective dose conversion factors in Tables B3, B4 or B5 can be used for external exposure. The differences between individual organ doses, and effective doses for external exposures are not large and use of the effective dose values in Tables B3, B4 or B5 for the non-stochastic DRL calculation is considered acceptable.

There are certain radionuclides (identified in Table C7, Appendix C) for which the DRL may be limited by the non-stochastic dose limit for skin. This dose rate for skin is described in Section 5.2 for external exposure. A separate DRL calculation should also be made for these radionuclides using the non-stochastic dose limit and the summation of all external exposure pathways in Equations 4.1 and 4.3 (for airborne discharges), Equations 4.1 and 4.6 (for surface water discharges) or Equations 4.1 and 4.9 (for groundwater discharges).

4.2.3 General Methodology for Calculating DRL for All Pathways

The calculation of DRL for simplified pathways leading from releases to atmospheric, surface water and groundwater environments has been illustrated in Section 4.2.1. For many applications, the algebraic expressions given in Section 4.2.1 may be appropriate for DRL calculations, while in other applications, these expressions may be inappropriate or incomplete. It is the responsibility of the user of this document to derive his own algebraic expressions for DRL calculations.

The general process involved in obtaining an algebraic expression for calculating the DRL for a radionuclide consists of the following steps.

1. Identify the environment to which the radionuclide is discharged, the exposure pathway(s), and the critical groups. Exposure pathways may differ for different radionuclides and for adults and infants;

2. Construct a pathway diagram for discharges to atmospheric, surface water and groundwater environments (analogous to Figures 4.2, 4.3 and 4.4 respectively);

3. Develop algebraic expressions for:
   $X_9(a)/X_S$ (4.3) (atmospheric discharges; see example Equation 4.2),
   $X_9(w)/X_S$ (4.6) (surface water discharges; see example Equation 4.5),
   $X_9(g)/X_S$ (4.9) (groundwater discharges; see example Equation 4.8),
   based upon Equation 4.1;

4. Select appropriate transfer parameter values. Transfer parameters are discussed in detail in Chapters 5, 6 and 7. Default values for transfer parameters between

various environmental compartments and for dose conversion factors are given in Appendices A and C.

In general, site-specific data or environmental models should be used to determine transfer values between various environmental compartments. Default values given in the appendix should be used only as a last resort; these values are often highly variable and the default values may not be representative of the specific pathway being assessed.

A sensitivity analysis to establish critical transfer parameters might aid in selecting the most critical pathways and parameters for specific study.

5. Calculate DRL based on the annual stochastic dose limit of 0.005 Sv. Calculations will be made separately for adults and infants.

6. Calculate DRL based on the annual non-stochastic dose limit of 0.05 Sv for those radionuclides listed in Table B2 (Appendix B). Calculations will be made separately for adults and infants.

7. Calculate DRL based on the annual non-stochastic dose limit for skin of 0.05 Sv for those radionuclides listed in Table C6 (Appendix C). The calculation will be the same as for adults and infants.

8. Select the smallest of the calculated DRL's in Steps 5, 6 and 7 above.

4.3 The Uranium-238 and Thorium-232 Decay Series

The radionuclides present in the effluents of uranium mines and mills consist of the U-238 decay series and the Th-232 decay series. These are shown in Figures 4.5 and 4.6, respectively (UNSCEAR, 1982). Radionuclides produced in less than one percent of the transmutation of the parent nuclide are not included.

The U-238 series may be divided into the following subseries in which the activity of the first member of the subseries controls, to a large degree, the activities of the decay products, and the half-lives of the decay products are relatively short:

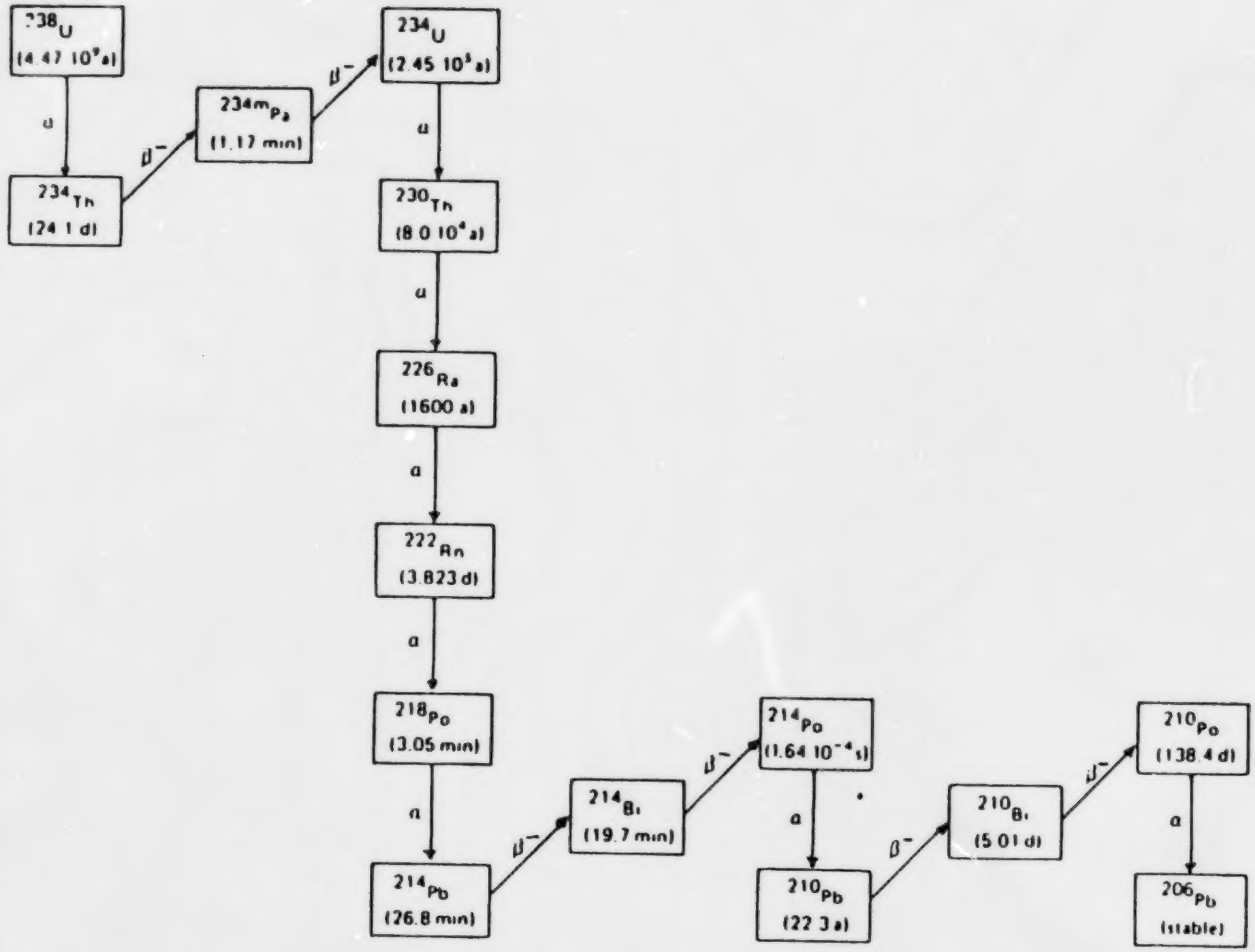
238U (4.47 10⁹ a)
α
234Th (24.1 d)
β⁻
234mPa (1.17 min)
β⁻
234U (2.45 10⁵ a)
α
230Th (8.0 10⁴ a)
α
226Ra (1600 a)
α
222Rn (3.823 d)
α
218Po (3.05 min)
α
214Pb (26.8 min)
β⁻
214Bi (19.7 min)
β⁻
214Po (1.64 10⁻⁴ s)
α
210Pb (22.3 a)
β⁻
210Bi (5.01 d)
β⁻
210Po (138.4 d)
α
206Pb (stable)

FIGURE 4.5 Uranium-238 decay series. Radionuclides produced in less than one percent of the transformations of the parent nuclide are not shown. (from Unscear 1982)

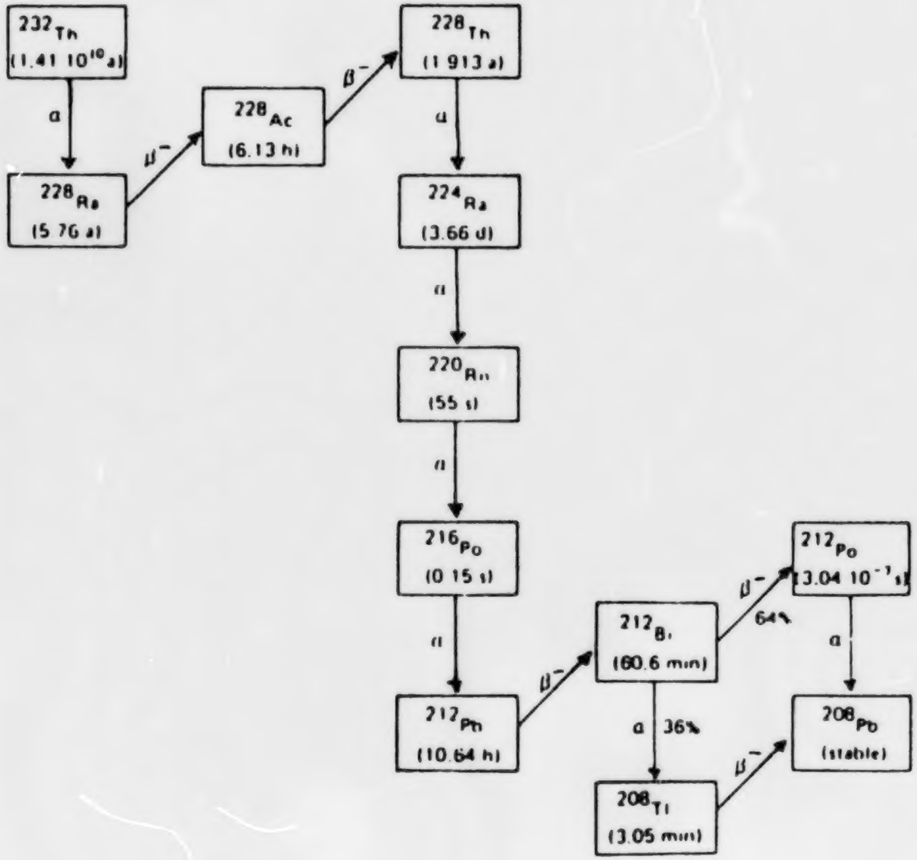
^{232}Th (1.41 10^{10} a)
α
^{228}Ra (5.76 a)
β$^-$
^{228}Ac (6.13 h)
β$^-$
^{228}Th (1.913 a)
α
^{224}Ra (3.66 d)
α
^{220}Rn (55 s)
α
^{216}Po (0.15 s)
α
^{212}Pb (10.64 h)
β$^-$
^{212}Bi (60.6 min)
β$^-$ 64%
α 36%
^{212}Po (3.04 10^{-7} s)
α
^{208}Tl (3.05 min)
β$^-$
^{208}Pb (stable)

FIGURE 4.6 Thorium-232 Decay Series (from Unscear 1982)

- U-238 → Th-234 → Pa-234m
- U-234
- Th-230
- Ra-226
- Rn-222 → Po-218 → Pb-214 → Bi-214 → Po-214
- Pb-210 → Bi-210
- Po-210

With the exception of the Rn-222 subseries, the radionuclides in each subseries may be regarded to be in secular equilibrium, i.e., all members in each subseries have equal activity. Derived Release Limits need only be calculated explicitly for the first members of the subseries.

Rn-222 is a noble gas and, due to its mobility, is often not in secular equilibrium with its decay products in effluents or in the environment. The equilibrium factor ϕ between Rn-222 and its decay daughter products is defined as follows:

$$\phi = \frac{\text{Potential alpha energy concentration of daughters}}{\begin{array}{c}\text{Potential alpha energy concentration of daughters}\\ \text{if they were in equilibrium with the parent Rn-222}\end{array}}$$

where the potential alpha energy of each daughter is the total alpha energy emitted during its decay to Pb-210. Potential alpha energy concentrations are given in units of $J \cdot m^{-3}$. Its relation to the older unit of Working Level (WL), as well as other special quantities and units associated with radon and its daughters, are given in Appendix F.

The Th-232 decay series may also be divided into subseries, as follows:

- Th-232
- Ra-228 → Ac-228
- Th-228 → Ra-224
- Rn-220 → Po-216 → Pb-212 → Bi-212 $\rightrightarrows$ Po-212 (64%)
  Tl-208 (36%)

Derived Release Limits are calculated explicitly for Th-232, Ra-228 and Th-228.

Rn-220 has a half-life of only 55 seconds, and a separate DRL explicitly calculated for it is not warranted. The impact of released Rn-220 and its daughters not maintained in equilibrium with their precursors (Th-228, Ra-224) is negligible relatively to the other radionuclides present in uranium mine/mill effluents. For example, for airborne releases assuming the nearest site boundary distance to be 1 km and a mean wind speed of $3\ m \cdot s^{-1}$, Rn-220 will have decayed by about six half-lives before reaching the boundary. At mine sites where thorium:uranium ratios are high, thorium may be extracted and stored separately from the tailings. Where this situation occurs and the critical group is situated very close to the mine site, doses from Rn-220 may require consideration.

Where Rn-220 is maintained in equilibrium with its precursors in some pathways, for example, the build-up of ground contamination from aerial deposition, its impact is accounted for in the DRL's for its precursors.

U-NAT and Th-NAT

It is common practice to measure and report natural uranium, U-NAT, contents or concentrations in mass units (typically ug/m^3, ug/kg, ug/L). It would be useful to establish a DRL for U-NAT expressed in mass units. Natural uranium consists of 99.275 atom % of U-238 in secular equilibrium with U-234 (0.0054 atom %), plus 0.72 atom % of U-235.

The specific activity (SA) of a radionuclide, in Bq/gm, is given by:

$$SA = 6.023 \times 10^{23}\ \lambda / AW \qquad (4.11)$$

where λ = radioactive decay constant (s^{-1});
AW = atomic weight;
6.023×10^{23} = Avogadro's number.

The specific activities of natural uranium radionuclides are calculated below.

Radionuclide	λ (s^{-1})	AW	SA (Bq/g)
U-238	4.91×10^{-18}	238.054	1.24×10^{4}
U-235	3.12×10^{-17}	235.044	8.0×10^{4}
U-234	8.96×10^{-14}	234.041	2.31×10^{8}
U-NAT	-	238.033	2.54×10^{4}

Note: All radionuclide data were obtained from Lederer et al. (1978).

It may be directly deduced from the above data and the composition of U-NAT that one microgram (ug) of U-NAT contains:

1.24×10^{-2} Bq U-238
1.24×10^{-2} Bq U-234
5.8×10^{-4} Bq U-235

Dosimetric calculations and DRL for U-NAT in terms of mass units (ug/a) will be based on the above relationships.

A similar approach for the DRL of Th-NAT expressed in mass units is not recommended. The principal thorium radionuclides in the natural thorium decay series are Th-232 (1.41×10^{10} a) and Th-228 (1.91 a). However, these two radionuclides are not always in secular equilibrium because the precursor of Th-228 is Ra-228 (5.76 a) which is a sufficiently long-lived radionuclide and chemically behaves differently from Th. Hence, the decay chain is effectively broken at Ra-228. Further, total thorium in uranium mine/mill effluents also contains Th-230 from the U-238 decay series. Radioisotopic analysis is, in general, required to determine Th-232, Th-228 and Th-230.

5.0 TRANSFER PARAMETERS FOR AIRBORNE RELEASES

Much of this section has been taken from the N288.1 report (CSA, 1984) with apppropriate modifications for radionuclides and pathways pertinent to the uranium mining and milling industry. Quotation marks indicate that the paragraph(s) enclosed by quotations are taken essentially verbatim from CSA (1984).

5.1 Atmospheric Dispersion (P_{S1})

5.1.1 Introduction

The construction and operation of a uranium mine/mill complex will introduce radioactive substances into the atmosphere. The movement of these radionuclides downwind, advection, and their dilution and dispersion cause their concentrations to decline with distance from the source. Predicting what radionuclide concentrations will be at the location of the critical group is dependent upon:

- the source type,
- the local topography,
- the local climate and wind patterns, and
- the distance from the critical group receptor to the source.

Mine/mill complexes will generally involve two types of sources:

- point sources or stacks, and
- area sources such as the pit or ore storage areas and tailings areas.

The releases from these sources are:

- particulate matter released either through stacks or mechanical handling of ore and waste rock;
- particulate matter released by the action of the wind on surfaces, such as waste rock, ore storage and tailings; and
- radon and its daughter products released from stacks, mechanical handling of ore and waste rock, tailings areas, ore storage piles or anywhere that the

operation increases the radon flux rate due to changing overburden characteristics.

The determination of source strength is beyond the scope of this document. However, some general comments on source strength estimation are warranted.

5.1.2 Source Considerations

The reader is cautioned that the approaches used in this section are not appropriate for calculations of short-term concentrations or levels on the site of the mine/mill complex.

Point source or stack emissions are generally considered to involve releases from any stack that has its exit at at least two and preferably two and one-half times the building height (Bowers et al., 1979). Plumes from stacks lower than this are influenced by surrounding building aerodynamics. This effect can cause localized elevated concentrations under certain circumstances. Stacks below two times the building height should be modelled using an approach similar to that provided in the ISC model (Bowers et. al., 1979). Point sources are definable in terms of:

- the height of the release (stack exit height),
- the temperature of the exit gas stream,
- the velocity of the exit gas stream, and
- the amount of material released.

The first three items are related to the effective height of release and are addressed in Section 5.1.3.2. These physical items are known or can be measured. Transfer parameters for each stack release will be defined in the next section.

For area sources, a different set of conditions governs the transfer parameter. Area sources tend to be diffuse, being the result of multiple small point sources closely congregated, or the result of generalized releases from moving sources. In the case of the mine/mill complex, each stack or building vent, provided one can define (a) a vertical velocity component, and (b) the exit height at at least twice the building height, is considered a stack or point source. However, open pit mining operations, wind erosion of tailings and storage piles, and vehicular traffic on the site result in area sources.

Generally, these sources are at or close to the general surrounding ground elevation. As such, one can neglect the height of the emission. This assumption was made on the basis of the naturally rugged terrain occurring around most existing or proposed mine sites in Canada. Should a site on a flat plain (where local topography does not vary as much as the various stockpiles do) be chosen, the source height should be considered. As the emissions are not strictly mechanically-driven, the buoyancy and the release velocity of the area source plume are not considered. For these reasons, it is best to consider area sources in terms of their proximity to each other, their relative strength (quantities of radionuclides released), and their orientation to critical receptors. Guidelines for this approach are presented in the area source transfer parameter discussion.

Line sources in the mine/mill complex are generally the result of vehicular traffic. Any vehicular operation within the confines of the mining area or within 2 km of the centroid of the mine area, may be considered as a contributor to the area source only. Vehicular traffic along roadways outside of this defined area must be considered on an individual basis. Normally, the effect of emissions of this type is limited to a short distance from the source (in the order of 1-3 km). Even under these circumstances, the normal source strength is low and the result of the emission is masked by the other sources in the area (IEC BEAK, 1984). Only if the critical group is within this distance from the road would it be necessary to include the term. At that time, the proponent would be required to utilize an appropriate algorithm, such as that in the PAL model (EPA-600/4-78-013), to calculate the transfer parameter. Since this case would be unusual, no further discussion of line sources will be presented.

5.1.3 Radionuclide Concentrations from an Isolated Point Source

5.1.3.1 General Transfer Parameter

The situation of point sources at the mine/mill complex is similar to that of point sources at a nuclear reactor facility. A detailed discussion of the calculations for point sources is presented in N288.1 (CSA, 1984), and is adapted for the following discussion.

"The following equations are recommended for calculating the long-term average concentrations of radionuclides in air at distances within a radius of about 20 km of the point of release. In most cases, the critical group is expected to reside within 20 km of the facility. For flat terrain and distances of 10 km or less, where local meteorological data are available, estimates of annual average concentration are accurate to within a factor of two (Hoffman, 1978).

The transfer parameter, P_{S1}, relating the concentration in the air, χ_1, at any point downwind of the source to the source release rate, $\chi_S(a)$, neglecting plume depletion due to radioactive decay and other removal processes (e.g., deposition and washout), is given by:

$$P_{S1} = \frac{\chi_1}{\chi_S(a)} \quad s \cdot m^{-3} \tag{5.1}$$

The expression for the long-term average transfer parameter from a continuous release is (Slade, 1968; Clarke, 1979):

$$P_{S1}(x, \theta_j, z) = \frac{\chi_1}{\sqrt{2\pi}\,\theta_j x} \sum_{i,k} \frac{f_{ijk}}{\sigma_{zi}\bar{u}_k} G(h, z, \sigma_{zi}) \tag{5.2}$$

where $G(h,z,\sigma_{zi}) = \exp\left[-\frac{(z-h)^2}{2\sigma_{zi}^2}\right] + \exp\left[-\frac{(z+h)^2}{2\sigma_{zi}^2}\right]$

θ_j = angle subtended by sector j (radians);

x = downwind distance from the source (m);

$\bar{u}_k$ = mean windspeed associated with windspeed interval k ($m \cdot s^{-1}$);

h = effective height of release (m) (see following section);

z = height above ground at which air concentration is calculated (m);

σ_{zi} = standard deviation of plume width in vertical direction associated with stability category i (m);

f_{ijk} = joint probability of occurrence of stability category i, with wind direction into sector j, and windspeed interval k.

The summation in Equation 5.2 is taken over all atmospheric stability categories (i) and windspeed intervals (k). The effect of mixing layer depths has not been included because the effect is small, within 20 km for most situations.

The parameters f_{ijk} should be determined from the maximum amount of relevant meteorological data.* It has been noted that a reliable analysis requires ten years of data (Alp and Lewis, 1985). ANS (1984) recommends five or more years of meteorological data for preliminary evaluations. Alp et al. (1985) indicate that for releases of ten hours duration or more, at least five years of data are required for developing representative concentration statistics. ANS (1984) notes that one year of on-site data is the absolute minimum amount of data that is representative, but stresses that three or more years is preferable. In the event that reliable local stability category data are not available, the long-term average dispersion characteristics may be estimated by using a site-specific wind rose and the default frequency distribution defined in Table C1.

The transfer parameter at ground level (z=0) is then given by:

$$P_{S1} = \frac{f_j n}{(2\pi)^{3/2} x} \sum_{i=1}^{6} \frac{f_i}{\sigma_{zi} u_i} G(h,\theta,\sigma_{zi}) \qquad (5.3)$$

where $G(h,\theta,\sigma_{zi}) = 2 \exp \left[- \frac{h^2}{2\sigma_{zi}^2} \right]$

---

*Such data are available for a large number of sites across Canada from Environment Canada, Canadian Climate Centre, 4905 Dufferin Street, Downsview, Ontario. The output of the "STAR" computer program provides monthly and annual frequency distributions of the six Pasquill stability classes distributed over 16 sectors of 22.5 degrees and six windspeed intervals.

and f_i = frequency of occurrence of stability category i (Table C1);

f_j = frequency with which the wind blows into sector j (from site wind rose);

n = number of sectors ($n = \frac{2\pi}{\theta_j}$)

The parameter $\bar{u}_i$ is the mean windspeed at a height of 10 m.

The concentration of the air pollutant is relatively insensitive to the stability category frequency within the normal range of the distribution across Canada and, hence, the values calculated using the default frequency distribution (Table C1) will, in most cases, be within 20% of those calculated using site-specific stability category data. Equation 5.3 is most sensitive to f_j.

The plume dispersion parameter, σ_{zi}, is a function of downwind distance, x, stability category, and the type of terrain. It can be calculated from the equations of Hosker (Hosker, 1974), Smith (Smith, 1972), Briggs (Briggs, 1974; Gifford, 1976), or Pasquill (Turner, 1970). The default values given in this document are derived from the equations of Hosker given in Appendix C, Tables C2, C3 and C4."

Default values of the parameter P_{S1} calculated from Equation 5.2 for uniform frequency of the wind into all sectors ($f_j = 1/n$), each frequency distribution class (Table C1), and a surface roughness length of 100 cm, are plotted in Figures A1-A6 (Appendix A), for various effective stack heights. The example roughness height was selected on the basis of the nature of most mining sites and their surroundings; these are generally in the boreal forest zone. A check of the changes in P_{S1} resulting from a surface roughness factor of 40 cm versus 100 cm showed a maximum difference of about 20%. Site-specific calculations may be required if surface characteristics do not match the forest type covering, for example, mines next to lakes with communities on the other side of the lake.

The value of P_{S1} may be estimated in the following manner:

- If the frequency with which the wind blows into the various sectors varies by less than a factor of two from the average, then P_{S1} may be taken directly from the appropriate stability category (Figures A1-A6).

- If the frequency with which the wind blows into the various sectors varies by more than a factor of two from the average, then the transfer parameter P_{S1} for each of the 6 stability categories may be obtained by multiplying the values in Figures A1-A6 by the ratio of the wind frequency into a particular sector to the wind frequency into that sector for a uniform wind rose.
- If no wind rose is available, then P_{S1} should be taken conservatively as three times the value taken from Figures A1-A6 for a uniform wind rose.
- The values for each stability category times the percent occurrence divided by 100 can then be summed for all stability categories to give the total weighted value of P_{S1}

5.1.3.2 Effective Height of Release

The height of release in the dispersion formula in Section 5.1.1 is the height above grade where turbulence in the free wind is the predominant cause of mixing. Several factors may operate to cause this height to differ from the physical height of the stack orifice. Excesses of buoyancy or momentum in the exit gases relative to the ambient air allow those gases to rise above the orifice before the plume bends over to assume a more or less horizontal trajectory. On the other hand, disturbance in the free flow of wind caused by the stack itself, or by neighboring buildings and other obstacles, may impart a downward component in the local wind field and the height where free dispersion begins is lower than the stack orifice.

Describing stack tip downwash or aerodynamic influences of surrounding buildings quantitatively has proven to be a formidable problem, and no generally applicable formula can be given. These difficulties are recognized in the definitions provided for point sources in the previous section of this report. A stack height of at least twice the building height will generally allow free dispersion provided there are no other buildings within 100 m of the source that are higher than the stack exit. The recommended ISC model (Bowers et al., 1979) incorporates a method of handling the aerodynamic influences of surrounding buildings.

This approach also allows for the situation of plume entrainment into the wake behind buildings. "When this occurs, a plume may reach the ground in a distance shorter than expected, if the stack was free-standing. Even so, the DRL's will not usually be limited by this circumstance except in the unusual situation where members of a critical group are present within the wake. The wake extends from the building a distance equal to no more than 20 building heights and is, therefore, in most cases, within 1 km of the source. The phenomenon typically occurs in wind speeds greater than about 5 $m \cdot s^{-1}$ and is a feature of neutral stability. Averaged over all weather conditions and all wind directions, the annual average concentrations at ground level at distances that are within the wake would not differ significantly from those experienced at the somewhat greater distances appropriate for a free-standing stack."

"On the other hand, conditions may exist which are propitious for a significant plume rise." All things considered, an acceptable default approach is to use the physical stack height as the release height.

A significant plume rise could justify the use of an appropriately higher DRL. While some process vents from the mine/mill complex may be at or only slightly above ambient temperatures, some sources have considerable thermal buoyancy, particularly when examined with respect to annual mean temperatures in the area.

Numerous models are available to describe plume rise. Fanaki (1979) reviews the models and notes the limitations; however, he suggests the Briggs (1969, 1971, 1972) formulae appear to provide better predictions than the others reviewed. The Atmospheric Environment Service of Environment Canada recommend the adoption of the Briggs formulae where possible (pers. comm., S.M. Daggupaty).

The Briggs formulae are discussed in detail in Appendix E. It is suggested that the effect be examined and plume rise be added to the stack height before applying Equation 5.2 or the default figures in Appendix A (Figures A1-A6).

5.1.4 Radionuclide Concentrations from Area Sources

As discussed earlier, many of the open operations at uranium mine/mill complexes result in a variety of open sources that introduce radionuclides into the atmosphere. Sources on the site should thus be reviewed to allow them to be grouped to provide reasonably regularly shaped areas of uniform source strength, and similar height to facilitate DRL calculations.

In the approach by the U.S. NRC for particulate emissions from area sources, it is recommended that the sources should be defined as square areas of no more than 300 m per side (NUREG/CR2011, 1981). This implies that multiple sources may be required to adequately describe some activities on-site. The average height of the "area" can then be utilized to describe the height of emissions above the receptor height. For negative heights, i.e., source lower than receptors, the height is conservatively set equal to zero.

For a continuous "area source", the proper procedure for obtaining the concentration of a gas or particulate at any point 'downwind' is to add the contributions from each infinitesimal point source in the area toward the concentration. This leads to the evaluation of an integral which, in general, poses difficulties for practical applications. For estimating concentrations at large distances (greater than two equivalent diameters of the source area; Raj and Kelelkar, 1974; Rausch et al., 1977), the following simplified approach suffices in most cases.

The area is replaced by a virtual point source of the same total source strength, but displaced upwind by a suitable distance (x_{vp}). This distance is calculated as a function of the source size and compensates for the apparent source spread provided by the area source.

While it is normal to calculate the displacement of the origin as a function of the horizontal dispersion coefficient, σy, the MILDOS model (NUREG/CR2011, 1981) provides a simplified formula for determining the virtual point source distance x_{vp} (See Appendix E.2). The displacement is calculated by:

$$x_{vp} = \frac{8}{\pi} S \qquad (5.4)$$

where S = length of the side of the area source;

X_{vp} = distance from center of area source to virtual point source.

The MILDOS documentation recommends, on the basis of U.S. NRC data, that the side of the area should be limited to a maximum of 300 m. The virtual source displacement is thus limited to about 800 m.

Utilizing the displacement distance defined in equation 5.4, the distance to the receptor from the source (x in equation 5.2) is redefined as x' where:

$$x' = x + x_{vp} \qquad (5.5)$$

Under some circumstances for receptor locations very close to the area source, a correction factor may be applied using the "narrow plume hypothesis" (see Appendix E). Use of this correction factor will result in a somewhat higher DRL.

The balance of the procedures discussed in Section 5.1.3.1 of this report, concerning the application of meteorologic factors and default values, apply for area sources.

5.2 External Dose from a Plume ($P(e)_{19}$)

"The exact calculation of the gamma dose rate in tissue from a finite plume of radioactive material is quite complex, and requires a knowledge of the radionuclide composition of the cloud. In order to simplify the calculations, the semi-infinite cloud model may be used under conditions where it will not lead to a significant under-estimation of the dose. The semi-infinite cloud model is conservative for releases, with an effective release height less than about 50 m at downwind distances greater than about 1 km." This situation is expected to apply at uranium mining and milling sites in Canada. At shorter distances for elevated releases, the semi-infinite cloud model may seriously under-estimate the dose and the finite cloud model must be used. This is discussed in more detail in Appendix D of CSA (1984) where equations are given for the finite cloud model. For calculating beta dose to skin and eye lens, the semi-infinite cloud model is applicable in all cases.

"The transfer parameter $P(e)_{19}$ relating the annual dose ($Sv \cdot a^{-1}$) to the annual average concentration of the radionuclide in air ($Bq \cdot m^{-3}$) is:

$$P(e)_{19} = (f_u + (1 - f_u)S) \quad (DCF)_a \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3 \tag{5.6}$$

where f_u = occupancy factor, or fraction of time an individual spends outdoors exposed to the unshielded cloud;

S = shielding factor, or fraction of the "out-of-door" dose that is received indoors due to shielding by buildings;

$(DCF)_a$ = dose conversion factor for a semi-infinite cloud ($Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3$).

Values of the shielding factor, S, for gamma radiation lie between 0.6 for a masonry house and 0.9 for a wood frame house (Burson and Profio, 1977). For purposes of calculating DRL, the default value is taken to be 0.9 when calculating effective dose. For the calculation of skin dose, S is taken to be 1.0, since for chronic emissions the air inside the house will be contaminated to the same extent as the outside air, and the short range of beta particles means that the air in the room will behave essentially as a semi-infinite cloud. The occupancy factor, f_u, will be site-specific; however, on average, an individual can be expected to spend about 20% of his time outdoors, leading to a default value for f_u of 0.2 (UNSCEAR, 1982).

Dose conversion factors $(DCF)_a$, are given in Table B3 (Appendix B)."

Radionuclides are considered whole body irradiators (from the gamma component) or external irradiators of the skin and lens of the eye (from the beta plus gamma components). In Table B3, "effective dose" is the sum of the weighted organ doses from gamma photons where the organ weighting factors used are those given in ICRP Publication 26 (ICRP, 1977). Values given in Table B3 are derived from Kocher (1983). Effective dose factors given by Kocher include contribution from skin dose weighted by a skin dose weighting factor of 0.01. Hence, the sum of organ weighting factors used in his calculations is 1.01. Skin dose contributions have been excluded from the effective dose factors given in Table B3 (Appendix B).

"The skin dose factors in Table B3 (Appendix B) are the sum of the skin gamma dose factors and the beta dose factors at a depth of 70 um below the body surface (Kocher, 1981). They will thus over-estimate dose to the eye lens, which lies at a depth of 300 $mg \cdot cm^{-2}$ (ICRP, 1975). The ratio of the skin (and eye) dose limit to the effective dose limit is ten (Section 2.2). Therefore, if the ratio of the skin dose factor to effective dose factor is less than ten, effective dose will be limiting."

Radionuclides for which the ratio of the skin dose factor to the effective dose factor exceeds ten are given in Table C6 (Appendix C). For these radionuclides, the skin dose should be summed for all external exposure pathways to ensure that the non-stochastic limit for skin is not exceeded (see general methodology in Section 4.2.3).

5.3 Deposition on Vegetated Soil (P_{13})

"The transfer parameter for deposition on vegetated soil from a radioactive plume under steady-state conditions is:

$$P_{13} = \frac{V_g}{\lambda_g} \quad m \qquad (5.7)$$

where V_g = deposition velocity ($m \cdot s^{-1}$);
λ_g = effective removal constant from the soil (s^{-1})."

Recommended values of V_g for airborne particulates are given in Table C7 (Appendix C) for different particle sizes. These values are sufficiently conservative to allow for the effects of both wet and dry deposition. Particulate radioactive materials released from the stack of facilities equipped with high efficiency particulate filters may be assumed to have sub-micron (< 1 um) particle sizes for which a deposition velocity of 0.003 m/sec applies. In most other situations, a deposition velocity of 0.01 m/sec is appropriate.

"The effective removal constant from the soil, λ_g, is governed by two processes - radioactive decay, characterized by the decay constant, λ_r, and environmental loss, characterized by a physical removal constant, λ_s. Therefore, $\lambda_g = \lambda_r + \lambda_s$. The constant, λ_s, is taken to be $2.20 \times 10^{-10}\ s^{-1}$ for all radionuclides. This corresponds to an upper limit of the physical half-life for removal from the soil of about 100 years (CEC, 1979).

Default values of the transfer parameter P_{13} are listed in Table A1 (Appendix A)."

5.4 Deposition on Forage, Crops, Browse and Lichen (P_{14})

"The transfer parameter for deposition from the plume to forage, crops and browse is:

$$P_{14} = \frac{V_g}{\lambda_e Y} \quad 1 - e^{-\lambda_e t_e} \quad m^3 \cdot kg^{-1} \tag{5.8}$$

where V_g = deposition velocity ($m \cdot s^{-1}$) (Section 5.3);
Y = vegetation yield ($kg \cdot m^{-2}$);
λ_e = effective removal constant from vegetation (s^{-1});
t_e = effective duration of the deposition (s).

The effective removal constant for agricultural forage and crops, and for natural vegetation (browse) (λ_e) is determined by radioactive decay (λ_r) and physical removal processes such as wind, rain and plant growth (λ_p), and is given by $\lambda_e = \lambda_p + \lambda_r$. The physical removal constant is taken to be 5.73×10^{-7} s^{-1}, which corresponds to a half-life for removal of 14 days (Hoffman and Baes, 1979). The deposition time, t_e, is taken as 30 days for pasture grass, and 60 days for other crops (USNRC, 1977)." The t_e for natural vegetation is taken to be 90 days, based on a maximum six-month period of exposed foliage divided by 2 for before consumption by game animals.

Default values of the vegetation yield, Y, are given in Table C8 (Appendix C).

Default transfer parameters P_{14} for forage grass ($Y = 0.28$ $kg \cdot m^{-2}$), vegetables ($Y = 1.4$ $kg \cdot m^{-2}$) and browse ($Y = 0.1$ $kg \cdot m^{-2}$) are given in Table A2 (Appendix A).

Deposition on Lichens

Lichens are a major link in the air-to-lichens-to-caribou-to-man food chain which may be a significant food chain in the vicinity of uranium mine/mill sites. Lichens derive their nutrition from soil via their fungal component, and also from the atmosphere. In the Precambrian Shield, where uranium mines and mills are located, soil development is poor and airborne materials are more important as a nutritional source. Lichens present high sorption areas, live for many years, and are very efficient accumulators of airborne particulate materials. For mining and milling areas, it is assumed that uptake from soil is negligible.

The physical removal half-life for particulate radioactive material deposited on lichens has been found to range from 2 to 17 years (Mattsson, 1975; Persson, 1971; UNSCEAR, 1977; Holm and Persson, 1975). A mean value of 5 years may be assumed, based on typical half-lives due to caribou grazing (Garner, 1972). The corresponding physical removal constant, λ_p, is 4.4×10^{-9} s^{-1}.

The recommended default value for the yield of lichens, Y, is 0.18 $kg \cdot m^{-2}$ dry weight, based on a standing crop biomass of 0.056-0.68 kg/m^{-2} fresh weight (mean 0.46) in Manitoba (Miller, 1976), and an assumed dry weight content of about 40% in lichens.

Lichens are long-lived. For the purpose of calculating DRL's, the effective deposition time, t_e, is assumed to be 10 years, or 3.15×10^8 seconds. This deposition time is based on an assumed operating life of 20 years for a uranium mine and a snow shielding factor of 2.

5.5 External Dose from Ground Deposits (P_{39})

"The transfer parameter P_{39} is given by:

$$P_{39} = f_r (f_u + (1 - f_u) S_g) \cdot (DCF)_g \qquad (5.9)$$

where $(DCF)_g$ = dose conversion factor for ground deposits (smooth infinite plane) $(Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^2)$;

f_u = fraction of time an individual spends outdoors exposed to the ground deposit;

f_r = dose reduction factor to account for non-uniformity of the ground surface;

S_g = shielding factor, or fraction of the out-of-door dose received indoors, reduced due to shielding by buildings.

The default value of f_u is 0.2 (Section 5.2) and the value of f_r is taken to be 0.7 (Burson, 1977). S_g is taken to be 0.4 for gamma radiation (Burson, 1977)." This value is conservatively applied to mixed beta gamma radiation. The default value of P_{39} is, therefore, equal to:

$$(0.7)\ ((0.2) + (0.8)(0.4))\ \ (DCF)_g = 0.36\ (DCF)_g$$

Dose conversion factors $(DCF)_g$ are given in Table B4 (Appendix B).

5.6 Inhalation by Humans ($P(i)_{19}$)

"The transfer parameter $P(i)_{19}$ relates the dose from inhalation of radioactive material ($Sv \cdot a^{-1}$) to the concentration in air ($Bq \cdot m^{-3}$). It is given by:

$$P(i)_{19} = I \cdot (OF)_i \cdot (DCF)_i \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3 \tag{5.10}$$

where I = inhalation rate ($m^3 \cdot a^{-1}$);

$(OF)_i$ = occupancy factor, or fraction of time an individual is exposed to the radioactive material in air;

$(DCF)_i$ = dose conversion factor for inhalation ($Sv \cdot Bq^{-1}$).

A comprehensive review of lung parameters as functions of age and sex can be found in pages 338-348 of ICRP-23 (ICRP, 1975).

Recommended values of I are:

Adult - $8.4 \times 10^3\ m^3 \cdot a^{-1}$

Infant (1-year old) - $1.4 \times 10^3\ m^3 \cdot a^{-1}$

In the absence of site-specific survey data, the default value of $(OF)_i$ is to be taken as 1. Dose conversion factors $(DCF)_i$ are listed in Tables B1 and B2 (Appendix B) for adults and infants (Johnson and Dunford, 1983)."

Radon and Radon Daughters

The radiological impact of radon releases is due primarily to the inhalation of radon daughters. Special quantities and units used for radon daughters are given in Appendix F. The dose conversion factors for the inhalation of radon and thoron daughters, $(DCF)_{Rn\ daughters}$, in Table F1 (Appendix F) are given in units of $Sv \cdot a^{-1}$ per $J \cdot m^{-3}$, where the daughter concentrations are expressed in terms of the potential alpha energy from daughter decay in units of $J \cdot m^{-3}$. Hence, for radon daughters, the transfer parameter $P(i)_{19}$ is given by:

$$P(i)_{19} = (OF)_i \cdot (DCF)_{Rn\ daughters} \quad Sv \cdot a^{-1} \cdot J^{-1} \cdot m^3 \qquad (5.11)$$

where $(OF)_i$ is the occupancy factor as defined in Equation 5.10.

It has been shown in Appendix F that $1\ J\ m^{-3} = 1.78 \times 10^8\ Bq\ m^{-3}$ of Rn-222 gas in radioactive equilibrium with its short-lived daughters. Hence:

$$(DCF)_{Rn} = 5.6 \times 10^{-9}\ \emptyset \cdot (DCF)_{Rn\ daughters} \qquad (5.12)$$

where $(DCF)_{Rn}$ is the dose conversion factor for Rn-222 gas in units of $Sv \cdot a^{-1}$ per $Bq \cdot m^{-3}$ of Rn-222 and $\emptyset$ is the equilibrium factor between Rn-222 and its decay daughter products, defined as follows:

$$\emptyset = \frac{\text{Potential alpha energy concentration of daughters}}{\text{Potential alpha energy concentration of daughters if they were in equilibrium with the parent Rn-222}} \qquad (5.13)$$

The transfer parameter $P(i)_{19}$ for Rn-222 gas is given by:

$$\begin{aligned} P(i)_{19} &= (OF)_i \cdot (DCF)_{Rn} \qquad (5.14) \\ &= 5.6 \times 10^{-9}\ \emptyset \cdot (OF)_i \cdot (DCF)_{Rn\ daughters} \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3 \end{aligned}$$

The value of ϕ selected should allow for the radon daughters released together with radon in mine vents and mill exhausts, as well as radon daughters grown in from radon during transit to receptor locations. The default value of ϕ is 1. Use of this default value will ensure that the DRL calculated for Rn-222 gas, based on equation 5.14, will be sufficiently conservative for all combinations of Rn-222 and its short-lived daughter products and that a separate DRL for potential alpha energy is not necessary. However, if desired, the DRL for potential alpha energy may be calculated using equation 5.11.

5.7 Plant Uptake from Soil (P_{34})

"The transfer parameter P_{34} relates the concentration of a radionuclide in vegetation ($Bq \cdot kg^{-1}$) to that deposited on the surface of the soil ($Bq \cdot m^{-2}$). It is given by:

$$P_{34} = \frac{B_v}{d} \; m^2 \cdot kg^{-1} \qquad (5.15)$$

where B_v = concentration factor for radionuclide in vegetation

$= \frac{\text{concentration in vegetation } (Bq \cdot kg^{-1});}{\text{concentration in soil } (Bq \cdot kg^{-1})}$

d = effective soil density ($kg \cdot m^{-2}$)."

The soil density, d, is taken to be 240 $kg \cdot m^{-2}$ for agricultural soil (USNRC, 1977) corresponding to a 15 cm plough depth. For uncultivated soils supporting natural vegetation, a 10 cm mixing depth is assumed, and the soil density is taken to be 160 $kg \cdot m^{-2}$. Default values for B_v are given in Table C9 (Appendix C) and for the transfer parameter P_{34} in Table A3 (Appendix A).

The values are divided into three classes:

- forage grass consumed by livestock where the parameter is the ratio of the radionuclide concentration in forage (dry weight) to that in soil;
- vegetables and fresh produce consumed by humans where the parameter is the ratio of the concentration in vegetation (fresh or wet weight) to that in soil; and

- browse that is consumed by moose where the parameter is the ratio of the radionuclide concentration in browse (dry weight) to that in soil.

Uptake of radioactive material from soil by lichens is negligible. The value of P_{34} for lichens may be taken to be zero.

5.8 Transfer from Vegetation to Animal Produce (P_{45}) and Aquatic Macrophytes to Moose (P_{75})

"The transfer parameter relating the concentration of radionuclides in animal produce to that in the food consumed by the animal is:

$$P_{45}, P_{75} = k_w \cdot Q_f \cdot F \quad \mathrm{kg \cdot kg^{-1}} \qquad (5.16)$$

where Q_f = feed consumption of the animal ($\mathrm{kg \cdot day^{-1}}$, dry weight);
F = fraction of animal's daily intake by ingestion which appears in each kg of produce ($\mathrm{day \cdot kg^{-1}}$);
k_w = non-grazing factor which allows for the fact that open grazing or growth of animal feed is not practised in winter in Canada."

Food consumption rates, Q_f, in terms of dry weight of forage for various animals are listed in Table C10 (Appendix C). Values of the parameter F for transfer to milk, beef, pork, eggs, poultry, and moose and caribou are given in Table C11. Moose and caribou should be assumed to consume browse and lichen, respectively, from areas that are relatively close to contaminant sources and are also within the recognized ranges of local game populations. Otherwise, specific studies on feeding habits should be carried out.

The winter non-grazing factor k_w is taken to be 1.0 for radionuclides with half-lives longer than one month. This allows for the fact that longer-lived radionuclides will not decay to any appreciable extent on animal feed, browse or lichen over winter.

Default values for the parameters P_{45} and P_{75} are given in Table A4 (Appendix A).

5.9 Transfer from Atmosphere to Animal Produce (P_{15})

"The transfer parameter from the atmosphere directly to animal produce, due to inhalation of radioactive material by animals, is given by:

$$P_{15} = I_a \cdot F' \quad m^3 \cdot kg^{-1} \tag{5.17}$$

where I_a = breathing rate of animal ($m^3 \cdot day^{-1}$);
F' = fraction of animal's daily intake by inhalation which appears in each kg of produce ($day \cdot kg^{-1}$)."

Breathing rates, I_a, of domestic animals have been estimated by Dr. H. Bayley (Dept. of Nutrition, University of Guelph, pers. comm., 1985). I_a values for moose and caribou have been estimated using the methodology developed by Ontario Hydro (1983), based on a human model. Breathing rates are listed in Table C12 (Appendix C). "Since little data are available on the transfer of inhaled material to animal produce, the values of F' have been estimated using human metabolic data (NRPB, 1979) and are summarized in Table C13 (Appendix C). Using these values, the transfer parameters P_{15} have been calculated and are listed in Table A5 (Appendix A)."

Unlike livestock, game animals such as caribou and moose range over wide areas; therefore, a long-term average radionuclide concentration in the air inhaled by these animals cannot be readily determined. If wild game consumption is significant in the critical group diet, it is recommended that the wild game be assumed to remain in the area of highest atmospheric radionuclide concentrations within the area where the species are known to occur. Alternatively, wild game distributions should be carefully studied to determine an "average" receptor location.

The fractional transfer of inhaled material to animal produce is given by (Ontario Hydro, 1983):

$$F' = \frac{f_a}{f_1} F \tag{5.18}$$

where F' = fraction of animal's daily intake by inhalation which appears in each kg of product ($day \cdot kg^{-1}$);

F = fraction of animal's daily intake by ingestion which appears in each kg of product ($\mathrm{day \cdot kg^{-1}}$);

f_a = fraction of inhaled activity reaching body fluids in man;

f_1 = fraction of ingested activity reaching body fluids in man.

Values of f_a are estimated based on Ontario Hydro (1983) using the ICRP lung model for reference man (ICRP, 1979), and f_1 values in Table B1 (Appendix B) and are given below:

Element	Solubility Class	f_1	f_a
U	Y	0.05	0.083
Th	Y	0.03	0.074
Ra	W	0.2	0.18
Pb	D	0.2	0.44
Po	W	0.1	0.15

5.10 Human Intake of Vegetation (P_{49}) and Animal Produce (P_{59})

"The transfer parameters P_{49} and P_{59} relating dose to ingestion of vegetable and animal produce are given by:

$$P_{49}, P_{59} = g_f \cdot I_f \cdot (DCF)_f \quad \mathrm{Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg} \tag{5.19}$$

where I_f = consumption rate of the appropriate foodstuff ($\mathrm{kg \cdot a^{-1}}$);

g_f = fraction of the consumed food arising from the contaminated source ($0 \leq g_f \leq 1$);

$(DCF)_f$ = dose conversion factor for ingestion exposure ($\mathrm{Sv \cdot Bq^{-1}}$).

Human consumption rates of vegetation and animal produce, to be used in the absence of site-specific surveys, are given in Table C14 (Appendix C). These values were selected based on a critical review of diet data (Nutrition Canada, Agriculture Canada; U.S. NRC, 1977, 1982; Baker, 1977; Zach, 1978; Beaton, 1980). Milk consumption rates for the infant were derived from ICRP-23 (ICRP, 1975). Consumption rates given for the average adult are also consistent with average annual "disappearance" of food in Canada. Disappearance is the sum of domestic production consumed in Canada and imports to Canada. Since disappearance includes spoilage and waste, it tends to slightly overestimate average consumption.

The "maximum" consumption rates in Table C14 represent the diet of exceptional individuals who may be considered to constitute the "critical group". The maximum consumption approach differs from the use of average consumption to calculate DRL's for nuclear facilities (CSA, 1984). Uranium mines and mills in Canada are generally located in areas that are remote from major population centres, and members of critical groups are more likely to have exceptional diets that may include much higher proportions of wild or natural food items (e.g., moose, caribou, fish and wild rice) than the average Canadian diet. A diet based on maximum consumption rates should normally be used in DRL calculations unless the identified group derives essentially all its diet from food grown in the immediate vicinity of the mine-mill facility. In the latter case, average consumption rates should be used since the same individuals would not be expected to consume the maximum of each food group.

Dose conversion factors $(DCF)_f$, provided by Johnson and Dunford (1983) and Dunford (pers. comm., 1985), are given in Tables B1 and B2 (Appendix B). The default value of g_f is taken to be 1 in the absence of site-specific data."

Default values for consumption of caribou and moose by the critical group (Table C15) are 25% and 50%, respectively, of the maximum consumption rates for other red meats. A lower value is selected for caribou than for moose, because caribou are migratory and may be unavailable to the critical group for a significant portion of the year. These default values are not based on survey data on the dietary habits of people living in remote areas of Canada; the available data show that the food consumption patterns of such populations vary greatly (Health and Welfare Canada, Food Directorate, pers. comm.). Because of this great variability, the user is urged to collect site-specific dietary data for the critical group.

6.0 TRANSFER PARAMETERS FOR RELEASES TO SURFACE WATERS

Much of this section has been taken from the N288.1 report (CSA, 1984), with appropriate modifications for radionuclides and pathways pertinent to the uranium mining and milling industry. Quotation marks indicate that the paragraph(s) enclosed by quotations are taken essentially verbatim from CSA (1984).

6.1 Dispersion of Liquid Effluents

6.1.1 Surface Water Transport

The transfer parameters P_{S2} relating surface water concentration X_2 ($Bq \cdot L^{-1}$) at a given location to the release rate $X_S(w)$ ($Bq \cdot s^{-1}$) is given by:

$$P_{S2} = \frac{X_2}{X_S(w)} \; s \cdot L^{-1} \qquad (6.1)$$

Models useful for estimating P_{S2} are given in Appendix D. Selection of the appropriate model depends upon the time scale and spatial scale of concern, the mixing regime, and the complexity of the model justified by available or measureable data. Three sample models can be used to describe many aquatic systems: the PFR (plug flow reactor), the DPFR (dispersed plug flow reactor), and the CSTR (chemically stirred tank reactor). The terminology is that of chemical engineering. In limnological applications, the CSTR model is also called 'well mixed'. The PFR and CSTR models are limiting cases for the DPFR model. The DPFR model accounts for advection, dispersion and removal mechanisms. The PFR model assumes that dispersion (diffusion) is negligible, whereas the CSTR model assumes that the rate of dispersion (mixing) is infinite. In the case where the mine operator is willing to base DRL calculations on radionuclide concentrations found in the liquid effluent, a model for P_{S2} is not required.

It is recommended that of the models given in Appendix D, P_{S2} can be evaluated as follows:

o for a riverine situation, the PFR model may be used. It is:

$$P_{S2} = \frac{1}{Q} \exp(-k\,\bar{t}_{PFR}) \qquad (6.2)$$

where Q = average annual advective flow rate past the point of concern, $L \cdot s^{-1}$

$\bar{t}_{PFR}$ = V_{PFR}/Q = hydraulic detention time between discharge point and point of concern, s

V_{PFR} = water volume of PFR between discharge point and point of concern, L

k = overall removal rate coefficient, s^{-1}

For example:

$$k = gf/\bar{z} + \lambda_r$$

where: λ_r = radionuclide decay rate constant, s^{-1}

g = settling rate of particulate forms of radionuclide, $m \cdot s^{-1}$

f = ratio of particulate to total concentration of radionuclide in water column

$\bar{z}$ = mean depth of waterbody between discharge point and point of concern, m

o for small to medium sized lakes, the CSTR model may be used. It is:

$$P_{S2} = \frac{1}{Q(1 + k\,\bar{t}_{CSTR})} \qquad (6.3)$$

where: $\bar{t}_{CSTR}$ = V_{CSTR}/Q = hydraulic detention time of well mixed water body, s

V_{CSTR} = volume of well mixed water body, L

o for a tidal mixing region, or the offshore zone of medium to large lakes (e.g., Lake Ontario), the DPFR model may be used. It is:

$$P_{S2} = 1\ (Q\sqrt{1 + 4\,kD/v^2})\ \exp((v/2D)(1 - \sqrt{1 + 4\,kD/v^2})\,x) \qquad (6.4)$$

Here: D = dispersion (or diffusion) coefficient, $m^2 \cdot s^{-1}$

v = advective flow velocity, $m \cdot s^{-1} = Q/A$

A = vertical cross-sectional area across which Q moves, m^2

x = distance from discharge point to point of concern, m

In the above equations, the transfer function is given in terms of the hydraulic residence time for the riverine (t_{PFR}) and for the lake (t_{CSTR}) systems. The hydraulic residence time is a fundamental property of water bodies. It can be related to physical distance, as follows. For river, the model may be redefined as:

$$P_{S2} = \frac{1}{Q} \exp\left[-k \frac{x_{PFR}}{v}\right] \tag{6.5}$$

where: x_{PRF} is the distance from the discharge point to the point of concern (m).

For the lake, the model may be redefined as:

$$P_{S2} = \frac{1}{Q + k A_S \bar{z}} \tag{6.6}$$

where: A_S = surface area of the lake (m^2)

For morphometric parameters (volume, mean depth, surface area) and flow variables (Q), default values cannot be provided; site-specific data are required to model dispersion. The hydraulic detention time of a river, t_{PFR}, or lake, t_{CSTR}, can be calculated as the volume of the river section or lake divided by the flow rate. Average annual flow rates can be estimated as the watershed yield (e.g., $m^3 \cdot s^{-1} \cdot ha^{-1}$) times the watershed area (ha). Yield and watershed area can be estimated from hydrological atlases and topographic maps, respectively.

Uncertainty associated with selection of the dispersion coefficient D is considerable. Values for the Great Lakes are typically of the order of 0.01-0.1 $m^2 \cdot s^{-1}$ (see Appendix D). If adequate information is not available, a default value of zero for D may be conservatively used, in which case Equation 6.2 should be substituted for Equation 6.4.

Values for g given in the literature have a considerable range. Settling rates for particulate forms of radionuclides have rarely been measured. Values for g for other substances provide guidance, assuming that adsorption into suspended particulates (O'Melia et al., 1983) is the main removal mechanism. Typical values range from 2.5×10^{-7} to 1.4×10^{-5} $m \cdot s^{-1}$ in the Great Lakes (Thomann and Ditoro, 1983). In the absence of good data for describing removal from the water column, a default value for g of zero can be used. This is a conservative approach, provided that water column pathways, rather than sediment pathways, provide the majority of the dose rate to the critical group.

Values for f, the parameter relating concentration of particulates to total concentration, range from 0 to 1. This parameter may be measured or modelled. Models for describing f typically take the form of:

$$f = \frac{K_D M}{\varepsilon} \qquad (6.7)$$

for adsorbed forms (see Equation 3, Appendix D). Here:

K_D is the linear distribution coefficient (e.g., units:volume/unit mass),
M is the concentration of aquatic particles (e.g., mass/unit volume), and
ε is the porosity.

Values for M and ε are site-specific. The distribution coefficient, K_D should not be confused with the sediment-water partition coefficient used in this report as the transfer parameter P_{28} (Section 6.7). K_D is the ratio of the concentration of radionuclide adsorbed onto suspended particulates, to the concentration in the liquid phase. K_D is typically measured in the laboratory. The partition coefficient between bottom sediment and water is that observed in the natural aquatic system; it expresses the ratio of a radionuclide concentration in bottom sediment to that in the overlying water. Bottom sediments may contain radionuclides not only due to sedimentation of particulates containing adsorbed particulates, but also due to direct adsorption from water onto the bottom sediment. Furthermore, radionuclides in bottom sediments may be remobilized and transported back to the overlying water column. Thus, the K_D and partition coefficient may differ considerably in a given body of water.

Values for K_D are also somewhat site-specific, although typical values measured in the laboratory have been used for other modelling studies (e.g., Snodgrass et al., 1982). Typical values measured for sediment concentrations (e.g., 1-10 g sediment/L) are tabulated in Table 6.1 for selected U-238 and Th-232 decay series radionuclides. A recommended set of default values is also provided for dispersion modelling purposes.

The evaluation of K_D at ambient suspended solids concentrations of surface waters (e.g., 1-10 mg/L) is, at present, a matter of considerable debate. There is an increasing amount of credible literature (e.g., initiated by O'Connor and Connelly, 1980) that K_D varies as a function of suspended solids concentration, although theoretically such a relationship should not exist. Typical relationships suggest that K_D increases by a half-order to an order of magnitude for an order of magnitude decrease in suspended solids concentration. At low solids concentrations, the K_D is independent of suspended solids concentrations. A sketch of the typical relationship is given in Figure 6.1. These measurements are performed in the laboratory under stirred conditions. Under non-stirred conditions, a few measurements suggest that lake sediments (several gm per dm^3) have K_D values which emulate those of dilute sediments (1 mg/dm^3) under stirred conditions.

For purposes of DRL calculations, it is suggested that the effects of removal by adsorption and sedimentation be checked by using literature values for g and estimates for K_D based upon sediment concentrations and K_D-suspended solids relationships. If removal is significant, experimental measurements of K_D should be made.

Settling of particulate forms of radionuclides can also occur if the radionuclides are incorporated into the matrix of particles (e.g., by algal growth, or coprecipitation into fines from tailings). An overall estimate for f for all particulate forms can be made if measurements are available.

6.2 Contamination of Land Vegetation by Spray Irrigation Water (P_{24})

"The transfer parameter relating the radionuclide concentration in vegetation to that in spray irrigation water is:

$$P_{24} = \frac{r \cdot L}{\lambda_e Y} (1 - e^{-\lambda_e t_e}) \quad L \cdot kg^{-1} \qquad (6.8)$$

TABLE 6.1: DISTRIBUTION COEFFICIENTS FOR RADIONUCLIDES, K_D (L/kg dry mass)

Element	K_D (freshwater)	K_D (marine)
U	16 - 270 (1)	3.2E5* (2)
	1E4* (2, 3)	5E2 (3)
Th	1 - 1E6 (1)	3.2E6* (2)
	5E3 (3)	5E4 (3)
	1E5* (6)+	
Ra	5E2* (3)	
	1.5E2 - 3.0E3 (4)	2E2 (3)*
	214 - 467 (5)	
Po	1E6* (6)	1E6* (2)
	1E3 (3)	1E3 (3)
Pb	1E3 (3)	1E3 (3)
	3.2E5* (2)	3.2E5* (2)
	3.6E4 (7)	

[1] Rancon (1973) (For Th, K_D is strongly pH - dependent).

[2] Santschi (1984)

[3] Snodgrass et al. (1982)

[4] Kirchmann et al. (1966)

[5] Serne (1974)

[6] Our estimate

[7] Stiller and Imboden (1985).

* Recommended default values for modelling purposes; these values were selected as being most representative (generally the mid-point of reported ranges or most recently cited values).

+ Assuming pH of water is near neutrality.

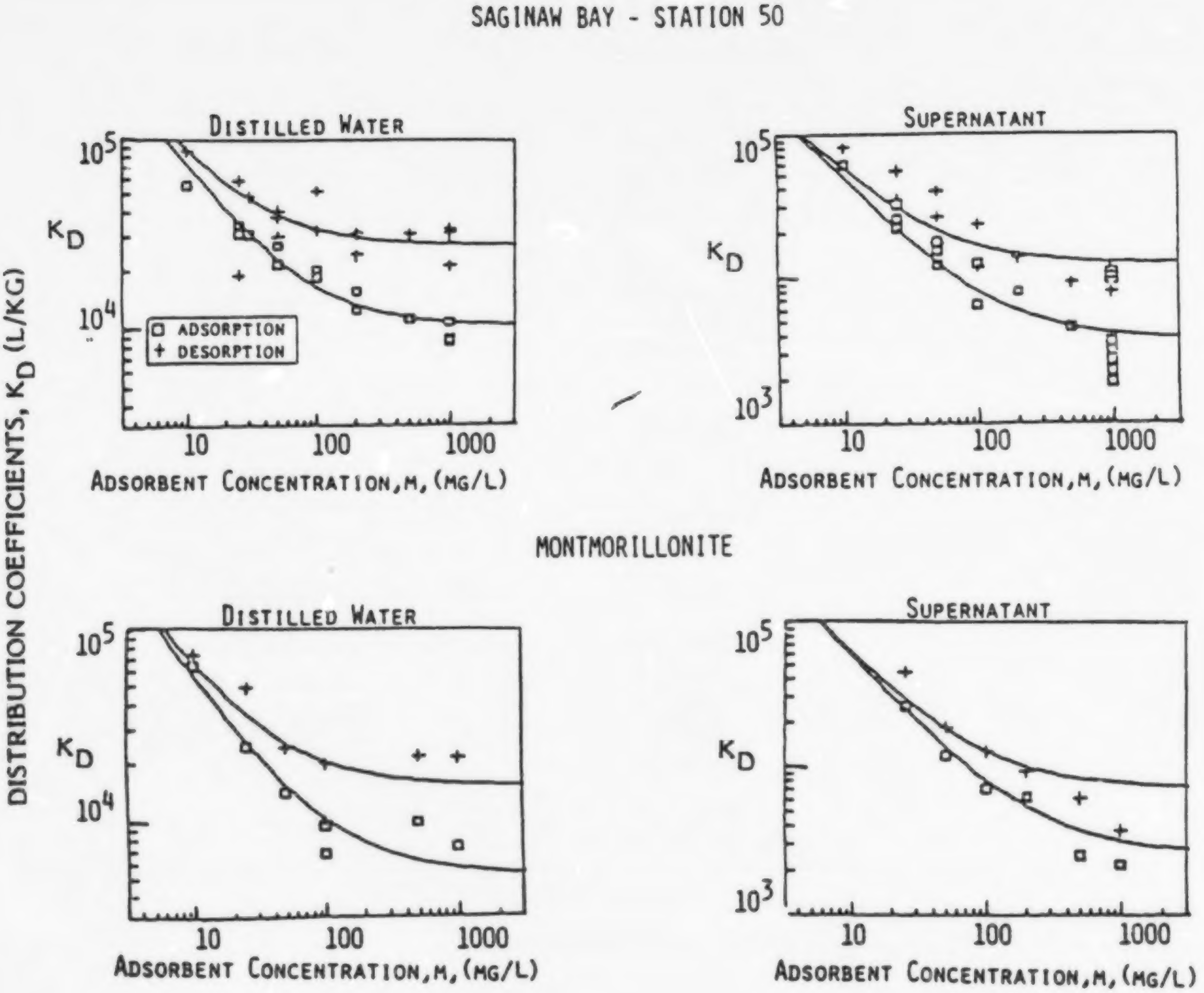

FIGURE 6.1: Adsorption and desorption distribution coefficient (K_D) variation with adsorbent concentration, Saginaw Bay Station 50 and montmorillonite. Distilled water and supernatant aqueous phases. Curves are for adsorption and desorption K_D's of a heavy metal onto respective solid phases (adapted from Ditoro et al., 1982).

where L = spray irrigation rate averaged over growing season ($L \cdot m^{-2} \cdot s^{-1}$);

r = initial fraction retained on vegetation (dimensionless);

λ_e = effective removal constant from vegetation (s^{-1});

Y = vegetation yield ($kg \cdot m^{-2}$, fresh weight);

t_e = effective duration of growing season(s).

The parameters λ_e, Y, t_e are assigned the same values as in Section 5.4; the initial fraction retained on vegetation, r, is taken as 0.05 (CEC, 1977). The spray irrigation rate, L, is taken to be about 2×10^{-5} $L \cdot m^{-2} \cdot s^{-1}$ averaged over the growing season (CEC, 1982). Default values of the transfer parameter P_{24} for leafy vegetables are given in Table A6 (Appendix A). Irrigation water is not considered a significant pathway for pasture in Canada." Neither are browse and lichens irrigated. Watering of backyard vegetable gardens, however, is applicable.

6.3 Contamination of Soil by Irrigation Water (P_{23})

"The transfer parameter from irrigation water to soil under steady-state conditions:

$$P_{23} = \frac{L'}{\lambda_g} \quad L \cdot m^{-2} \qquad (6.9)$$

where L' = annual average irrigation rate ($L \cdot m^{-2} \cdot s^{-1}$);

g = effective removal constant from the soil (s^{-1}).

The parameter λ_g has the same value as in Section 5.3., i.e., $\lambda_g = \lambda_r + \lambda_s$ where λ_r = radioactive decay constant and $\lambda_s = 2.20 \times 10^{-10}$ s^{-1}, corresponding to a removal half-life from the soil of 100 years. The annual average irrigation rate, L, is taken conservatively to be the same as the rate during the growing season, i.e., 2.3×10^{-5} $L \cdot m^{-2} \cdot s^{-1}$ (Zach, 1978). The resulting default values of P_{23} from Equation 6.9 are given in Table A7 (Appendix A)."

6.4 Water Intake by Land Animals (P_{25})

"The transfer parameter relating the concentration of a radionuclide in animal produce to that in the water consumed by the animal is:

$$P_{25} = k'_w \cdot Q_w \cdot f \quad L \cdot kg^{-1} \tag{6.10}$$

where Q_w = daily water intake of the animal ($L \cdot day^{-1}$);

F = fraction of the animal's daily intake which appears in each kg of produce ($day \cdot kg^{-1}$);

k'_w = fraction of the animal's annual water intake which arises from the contaminated source.

Water intake rates, Q_w, for various animals are given in Table C15 (Appendix C); values of F are listed in Table C12. The default value of k'_w is taken as 1." Moose and caribou should be assumed to drink from those surface waters that are most contaminated within the range of local populations, unless specific studies show that other water sources are used.

Default values of the parameter P_{25} are given in Table A8 (Appendix A).

6.5 Drinking Water Intake by Humans ($P(i)_{29}$)

"The transfer parameter relating dose to humans to drinking water intake is:

$$P(i)_{29} = \rho \cdot k''_w \cdot I_w \cdot (DCF)_f \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L \tag{6.11}$$

where $(DCF)_f$ = dose conversion factor for intake by ingestion ($Sv \cdot Bq^{-1}$);

I_w = drinking water intake ($L \cdot a^{-1}$);

k''_w = fraction of drinking water intake arising from the contaminated source;

ρ = removal factor to account for processes such as sedimentation and removal of radionuclides by water treatment plants.

Dose conversion factors, $(DCF)_f$, for the ingestion pathway are given in Tables B1 and B2 (Appendix B). Water intake has been reviewed by the ICRP (1975), and the total intake by reference man is 1.95 $L \cdot day^{-1}$, which is comprised of 0.3 $L \cdot day^{-1}$ of milk, 0.15 $L \cdot day^{-1}$ of tap water, and 1.5 $L \cdot day^{-1}$ from other sources (tea, coffee, juices, etc.). The value for infants (1-year old) is estimated from daily water balance: losses in urine (500 mL), losses in sweat (200 mL), "insensible" losses (300 mL), and gains in milk (600 mL), food

(50 mL), and by oxidation of food (100 mL). The balance gives a total water intake of 250 mL·day^{-1}. Certain groups of infants may be fed on reconstituted powdered milk adding an additional 600 mL·day^{-1} of water, in which case the total water intake would be 850 mL·day^{-1}. Therefore, on an annual basis, the parameter I_w is:

Adults:	700 L·a^{-1}
Infants drinking fresh milk:	90 L·a^{-1}
Infants drinking reconstituted powdered milk:	300 L·a^{-1}

The default value for k''_w is 1.

The value of ρ should be determined on a site-specific basis. Its default value is 1.0."

6.6 Biological Concentration in Aquatic Foods (P_{26}, P_{27})

6.6.1 Fresh Water Environment

"The transfer parameter P_{26}, commonly referred to as the bioaccumulation factor, is defined under equilibrium conditions as:

$$P_{26} = \frac{X_6}{X_2} = \frac{\text{Concentration in edible portion of Biota (Bq·kg}^{-1})}{\text{Concentration in Water (Bq·L}^{-1})} \qquad (6.12)$$

$$\text{Similarly, } P_{27} = \frac{X_7}{X_2} \ \text{L·kg}^{-1} \qquad (6.13)$$

The most important Canadian fresh water organism from the point of view of human consumption is fish. The bioaccumulation of radionuclides in fish is a complex process involving direct uptake from water, absorption from foods such as aquatic plants and other biota, and absorption from ingested sediments. Since the concentrations in these foods and in sediment are also related to that in water, transfer parameters such as P_{86}, P_{87}, P_{76} (see Figure 4.1) are not explicitly used in determining concentrations in fish.

The bioaccumulation factor for a given radionuclide may vary over a wide range, depending on the fish species and such factors as stable element concentrations, pH, temperature, nutrient levels, and the physico-chemical forms of the radionuclides. For the purposes of DRL calculation, default P_{26} values (Table A9, Appendix A) for fish flesh are recommended for use in the absence of site-specific data. These were selected based on a review of literature data."

For most elements, the bioaccumulation factor in fish is correlated with the dissolved mineral content of water; it generally decreases with increasing mineral content. In Table A9, separate values are given for uranium bioconcentration factors in waters of low and high mineral content. A total dissolved solids (TDS) concentration of 150 $mg \cdot L^{-1}$ is used as the lower bound for high mineral content water.

Another potentially important food item for the critical group is wild rice, harvested from an emergent aquatic macrophyte found in shallow portions of many small lakes in Canada. Although wild rice is harvested commercially to some extent, little is used in the average Canadian diet. However, in small communities having a hunting and gathering type of existence, such as many native communities, wild rice may form a more significant portion of the diet.

6.6.2 Marine Environment

"Recommended default bioaccumulation factors (P_{26} and P_{27}), in marine biota based on review of literature data, are given in Table A10 (Appendix A)."

6.7 Transfer Parameter for Surface Water to Aquatic Sediments (P_{28})

Removal of radionuclides from the water column by sedimentation, as outlined in Subsection 6.1.1, results in radionuclide accumulation in the sediments. Some of these contaminated sediments may be deposited along shorelines during high water or storm events. The critical group may be exposed to these sediments during such activities as sunbathing on the beach and fishing from the shoreline.

The transfer parameter P_{28} relates the concentration of radionuclide in sediment, X_8 ($Bq \cdot kg^{-1}$) to the concentration in water, X_2 ($Bq \cdot L^{-1}$) as follows:

$$P_{28} = X_8/X_2 \quad L\cdot kg^{-1} \tag{6.14}$$

Values of P_{28} selected from the literature for freshwater and marine sediments are given in Table AII (Appendix A). These P_{28} values are based on empirical observations of total concentrations in sediment, i.e., including both sediment solid and pore water components. These values may be derived for steady-state or non-steady-state conditions. Values based on steady-state are most applicable for calculating the DRL - such values may be derived by modelling physical transport processes (Appendix D). Use of these P_{28} values is conservative because shoreline sediments may be coarser and have less organic content than deep sediments, thereby having less capacity for radionuclide adsorption, and may also be "diluted" with uncontaminated deposits of terrigenous origin.

X_8 is defined as the activity per wet weight of sediments. Values for P_{28} (Table A.II) are given as activity per dry weight of sediments. These measurements are made by taking a known volume of wet sediment, drying the sediment and measuring the activity in the sediments. During the drying process, radionuclides contained in the pore water are incorporated onto the solid fraction. Hence, the measured solid fraction includes radionuclides on the solid and in the pore water before the measurement. A formula for relating wet weight to dry weight is given in Table AII.

6.8 External Dose from Contaminated Shoreline (P_{89})

"The transfer parameter P_{89} relating the external dose rate from contaminated shoreline sediment to the radionuclide concentration in the sediment is given by:

$$P_{89} = (OF)_s \cdot W \cdot (DCF)_g \cdot d_s \cdot (DF)_s \quad Sv\cdot a^{-1}\cdot Bq^{-1}\cdot kg \tag{6.15}$$

where $(OF)_s$ = shoreline occupancy factor, or fraction of time an individual spends over contaminated shoreline;

W = shore-width factor that describes the shoreline exposure geometry;

$(DCF)_g$ = dose conversion factor for a uniformly contaminated ground ($Sv\cdot a^{-1}\cdot Bq^{-1}\cdot m^2$);

d_s = effective density of sediment ($kg\cdot m^{-2}$);

$(DF)_s$ = dilution factor for shoreline deposits (≤ 1) which allows for non-equilibrium between suspended sediment and shoreline deposits.

It is assumed that exposed shoreline sediment is contaminated to some depth to a radioactivity concentration ($Bq \cdot kg^{-1}$) determined by the transfer parameter P_{28}. The depth of this contaminated layer is taken to be 0.025 m, which in conjunction with a dry sediment density of 1,600 $kg \cdot m^{-3}$ gives an effective sediment density, d_s, of 40 $kg \cdot m^{-2}$ (USNRC, 1977a). The dose contribution from radionuclides lying below this depth is ignored. The dose conversion factors $(DCF)_g$ for a uniformly contaminated plane are listed in Table B4 (Appendix B). They are the same factors as are used for ground contamination in the parameter P_{39} (Section 5.5).

The shoreline occupancy factor should be obtained from site surveys. In the absence of any commercial activity such as clam digging, the default value for $(OF)_s$ is 0.01, which corresponds to about 100 hours per year over contaminated shoreline. Recommended shore-width factors, W, are given in Table C16, Appendix C (USNRC, 1977a).

Because the source is distributed in the sediment, the bulk of emitted beta radiation is absorbed in the sediment. Skin dose from beta particles will thus be negligible compared to the gamma dose." Users of this document are cautioned that Equation 6.15 will greatly over-estimate skin doses for the pure beta emitters (or those radionuclides in which the beta dose dominates the gamma dose from a plane source).

"The value of $(DF)_s$ is highly site-specific. Its value may range from close to unity, for bottom sediments build up from direct sedimentation from the water column, to quite small values for shorelines subject to wave action. In the absence of site-specific measurements, its value should conservatively be taken to be unity."

6.9 Water Immersion Dose ($P(e)_{29}$)

"The transfer parameter relating dose from immersion in contaminated water to the concentration of a radionuclide in that water is given by:

$$P(e)_{29} = (DCF)_w \cdot (OF)_w \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L \qquad (6.16)$$

where $(DCF)_w$ = dose conversion factor for immersion in contaminated water ($Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$);

$(OF)_w$ = water occupancy factor; or fraction of time spent swimming.

In the absence of site-specific data, the default value of $(OF)_w$ is taken as 0.01, corresponding to about 100 hours per year. Values of $(DCF)_w$ are given in Table B5 (Appendix B)."

6.10 Human Intake of Aquatic Food (P_{69}, P_{70})

"The transfer parameter relating dose to intake of aquatic food is:

$$P_{69} = g_f \cdot I_f \cdot (DCF)_f \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg \qquad (6.17)$$

where I_f = consumption rate of aquatic food ($kg \cdot a^{-1}$);
$(DCF)_f$ = dose conversion factor for ingestion ($Sv \cdot Bq^{-1}$);
g_f = fraction of consumed aquatic food arising from the contaminated source."

Consumption rates of aquatic food (freshwater or marine) vary widely over regions of the country, so that use of Canadian average data, as recommended under certain conditions (Section 5.10) for consumption of vegetation and animal produce, is not recommended here.

For calculating DRL's, it is recommended that site-specific food consumption patterns for the critical group be determined. In lieu of site-specific data, very conservative default values based on one small portion of fish (0.125 kg) and one meal of wild rice (0.1 kg) per day are recommended.

Dose conversion factors, $(DCF)_f$, are given in Tables B1 and B2 (Appendix B). The default value of g_f is taken to be 1 in the absence of site-specific data.

6.11 Intake of Radon from Surface Water by Humans (P'_{29})

Radon dissolved in surface water can be released into the air in houses due to various domestic activities such as washing and showering and give rise to inhalation hazards. The transfer parameter P'_{29} is calculated in the same manner as given in Section 7.4 for P'_{09}.

7.0 TRANSFER PARAMETERS FOR RELEASES TO GROUNDWATER

7.1 Transfer Parameter for Liquid Effluents Discharged to Groundwaters (P_{S0})

The transfer parameter P_{S0} relating groundwater concentration X_0 ($Bq \cdot L^{-1}$) at a given location in the ground to the release rate $X_S(g)$ ($Bq \cdot s^{-1}$) is given by:

$$P_{S0} = \frac{X_0}{X_S(g)} \quad (s \cdot L^{-1}) \tag{7.1}$$

Models useful for describing P_{S0} are given in the appendix. Selection of the appropriate model depends upon the time scale and the spatial scale of concern, the amount of dispersive mixing which occurs, and the complexity of the model justified by available data. If data are insufficient, but manpower and economic resources can be justified, field measurements can be made to validate selection of a model and to provide information for specifying coefficients in the model.

Two simple models given in the appendix 'conceptually' describe most of the transport variations observed in groundwater systems - the PFR (plug flow reactor) model (Equation D3; Appendix D) and the DPFR (dispersed, plug flow reactor) model (Equation D1; Appendix D). The PFR is a simplification of the DPFR model in which dispersion is insignificant relative to advection (bulk groundwater flow).

In the mathematical form given in Appendix D, the models are most appropriate for surface waters - other processes such as adsorption become more significant in groundwater systems.

Including the effects of adsorption decay dispersion, and advective flow, the DPFR model gives:

$$P_{S0} = \frac{1}{Q\sqrt{1 + \frac{4k_D}{V^2} \frac{(1 + K_D \rho_B)}{\epsilon_G}}} \exp.\left[\frac{V}{2D}\left[\frac{1- \quad 1 + 4kD(1 + K_D \rho_B)}{V^2}\right]\right] \tag{7.2}$$

as a simple relationship for evaluating P_{S0}. The symbols are as defined in Section 6.1. Additional symbols include:

ρ_B = bulk mass density of ground (typical values of 1.6 g/cm^{-3})

ε_G = porosity of ground (typical values of 0.2 to 0.4)

For certain groundwater environments, the approximate transfer parameter for DRL purposes may be:

$$P_{50} = 0 \tag{7.3}$$

The DRL concept assumes steady-state conditions and that radionuclide discharge will reach humans during the operating lifetime of the facility. The travel time through groundwater environments is normally much longer than through surface water or atmospheric pathways. Adsorption processes increase the travel time of radionuclides relative to that for a conservative element (i.e. one which does not adsorb). The travel time of a radionuclide through a groundwater pathway (e.g. to a drinking water well, or lake) may be longer than the operating life of a tailings facility, in which case $P_{50} = 0$.

One can assess whether the groundwater pathway approaches steady-state by calculating the retardation factor (RF, see Appendix D, Equation D15). If the quantity:

$$\frac{V}{RF} \tag{7.4}$$

is larger than the operating life of the facility (where V is the groundwater velocity), the radionuclide may not reach the human food chain in any significant quantity. Hence, if the numerical value of V/RF is much greater than the operating life of the facility, P_{50} equals 0.

Evaluation of Equation 7.4 provides an approximation of whether or not significant quantities of a radionuclide could reach the food chain. To obtain a more precise estimate, one should consider the length of the flow path, the diffusion coefficient, the distribution coefficient and the radionuclide activity considered as significant. A sketch of these aspects is given in Figure 7.1. The mean position of the median value of the normalized concentration profile of a sorbing and of a non-sorbing radionuclide at any time 't' are given by 'a' and 'b' respectively in Figure 7.1.

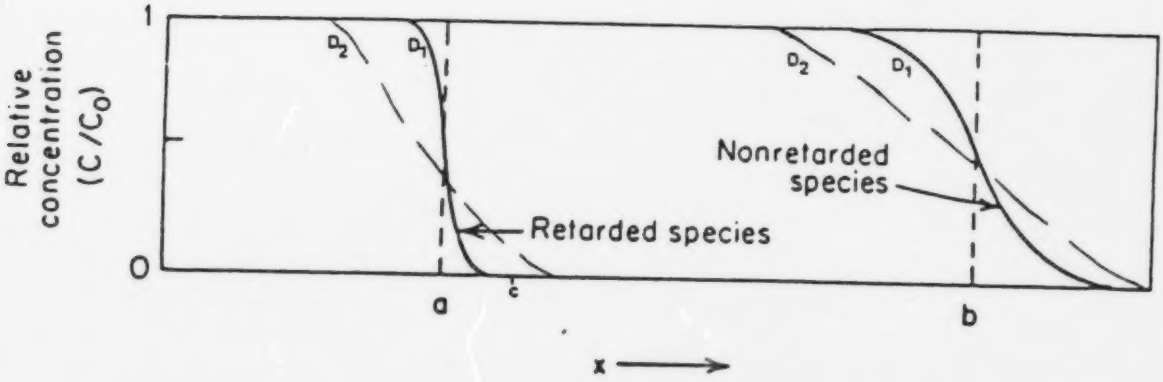

FIGURE 7.1: Relative position at any time 't' of adsorbed and nonadsorbed radionuclides travelling through a column of porous materials. Dotted lines 'a' and 'b' describe mean position of the 50% value of the normalized concentration profiles for adsorbed (retarded) and nonadsorbed (non-retarded) species. Profiles are for two different dispersion coefficients: $D_2 > D_1$. The front described by D_1 has not reached position 'c', whereas the front described by D_2 has passed by position 'c'.

The mean position is the basis of Equation 7.4. The relative magnitude of the distance characterized by position 'a' and position of 'b' is described by the retardation factor, while the absolute magnitude of position 'b' represents the distance that a non-sorbing radionuclide would travel in time 't' at velocity 'v'. But due to dispersion, part of the radionuclide advances ahead of the medium position of the front.

For example, consider the case where a household well is located at position 'c' (Figure 7.1) and where, over the operating life of a mine-mill facility, the medium position of a sorbing radionuclide is position 'a'. If the significant concentration of a radionuclide is 1% of C_o, one could conclude that the $P_{S0} = 0$ if the aquifer dispersion coefficient is D_1, because the concentration is less than 1% of C_o whereas one may conclude that $P_{S0} \neq 0$ if the aquifer dispersion coefficient is D_2. Further aspects for evaluating whether or not $P_{S0} = 0$ are given in Appendix D.

Once released to groundwater, radionuclides may be transported in groundwater to areas where release to surface water or atmosphere is possible. Release from groundwater to the atmosphere, however, is considered in the special case of well water used for domestic purposes, such as showering, as a potentially significant pathway for DRL calculations.

7.2 Transfer Parameter for Groundwater to Surface Water (P_{02})

The transfer parameter P_{02} relating surface water concentration to groundwater concentration (X_0) is:

$$P_{02} = \frac{X_2}{X_0} \tag{7.5}$$

A transfer function for describing P_{02} is dependent upon the nature of the groundwater-surface water pathway and the characteristics of the surface water. Examples of alternative possibilities for P_{02} include the following:

1. Water is pumped from a well, to a house and subsequently discharged to a lake. A certain fraction of the radionuclide (fr) is retained in the dwelling. Decay of the radionuclide in the house is insignificant. The receiving water body is a narrow river described as a PFR.

Then, $$P_{02} = (1-f_r) \frac{Q_p}{Q} \exp^{(-k\bar{t}_w)} \qquad (7.6a)$$

where, Q_p is the pumping rate from the well ($L \cdot s^{-1}$).

Q is the flow rate in the river ($L \cdot s^{-1}$).

k is the removal rate constant in the river (s^{-1}).

$\bar{t}w$ is the hydraulic detention time of the river (units: s) between the point of discharge and the physical location represented by X_2. $\bar{t}_w$ is equal to V_{PFR}/Q.

where, V_{PFR} is the volume of the river between the point of discharge and the physical location represented by X_2

2. The conditions are the same as in Example 1, but the receiving water body is a lake.

Then, $$P_{02} = \frac{(1-f_r)Q_p}{Q} \frac{1}{1 + k\bar{t}_w} \qquad (7.6b)$$

where the symbols are as defined above except:

Q is the flow rate out of the lake ($L \cdot s^{-1}$)

$\bar{t}w$ is the hydraulic detention time(s) of the lake, equal to V_{CS}/Q

where, V_{CS} is the volume of the lake (L)

3. Groundwater is discharged through the shoreline to a lake.

Then, $$P_{02} = \frac{V_c A_G}{Q} \frac{1}{1 + k\bar{t}_w} \qquad (7.6c)$$

Here the symbols are as defined for Example 3.

and V_c is the mean velocity of the radionuclide in the ground ($m \cdot s^{-1}$)

A_G is the cross-sectional area (m^2) through which groundwater flows into the lake.

7.3 Contamination of Land Vegetation by Spray Irrigation Using Groundwater (P_{04})

This parameter is calculated in the same manner as given in Section 6.2 for P_{24}.

7.4 Contamination of Soil by Irrigation from a Groundwater Source (P_{03})

This parameter is calculated in the same manner as given in Section 6.3 for P_{23}.

7.5 Groundwater Intake by Land Animals (P_{05})

This parameter is calculated in the same manner as given in Section 6.4 for P_{25}. Use of groundwater as a drinking water source would apply to livestock. Where both surface water and groundwater provide drinking water sources, total intake must not exceed the Q_w values given in Table C15. Moose and caribou would consume contaminated groundwater only following discharge to surface water.

7.6 Intake of Groundwater by Humans ($P(i)_{09}$)

This transfer parameter is calculated in the same manner as given in Section 6.5 for $P(i)_{29}$.

7.7 Intake of Radon from Groundwater by Humans (P'_{09})

Radon dissolved in groundwater can be released into the air in houses where groundwater is used, thus giving rise to inhalation hazards. Release could be due to various domestic activities such as washing and showering. The transfer parameter, P'_{09}, which relates the dose from inhalation of radon daughters to radon concentration in groundwater, is given by:

$$P'_{09} = \frac{(X_1)_{Rn}}{(X_0)_{Rn}} = K_{Rn} \cdot P(i)_{19} \quad Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L \qquad (7.7)$$

where K_{Rn} = Rn-222 air-to-water concentration ratio ($Bq \cdot m^{-3} \cdot Bq^{-1} \cdot L$);

$(X_0)_{Rn}$ = Rn-222 concentration in groundwater ($Bq \cdot L^{-1}$);

$(X_1)_{Rn}$ = Rn-222 concentration in air resulting from release from groundwater ($Bq \cdot m^{-3}$);

$P(i)_{19}$ = transfer parameter for human inhalation of Rn-222 gas as given by Equation 5.15, Section 5.6.

Values of K_{Rn} given in UNSCEAR (1982), converted to units of $L \cdot m^{-3}$, vary from 0.01 to 1, with the higher values observed while taking warm water showers. An average value of 0.2 $L \cdot m^{-3}$ is recommended.

7.8 Water Immersion Dose ($P(e)_{09}$)

In most cases, immersion in water taken from a groundwater source is not expected to contribute significantly to the total dose. In the house, showering may result in exposure to radon, as outlined in Section 7.7.

Immersion in bath water or in a swimming pool filled with groundwater may result in a dose to the bather. In this case, the dose is calculated as described in Section 6.9. A default groundwater occupancy factor $(OF)_g$ equal to the surface water occupancy factor $(OF)_w$, or 0.01 (see Section 6.9) is reasonable for the bathing or swimming pool scenario.

8.0 APPLICATION GUIDES

Subsections 8.1.1, 8.1.2 and 8.2 have been taken largely from corresponding subsections of the N288.1 Standard (CSA, 1984). The paragraphs enclosed by quotation marks are taken verbatum or with very minor revision from CSA (1984).

8.1 Source Averaging Time

8.1.1 Airborne Releases

"The calculation of DRL's for airborne releases assumes a uniform source emission rate and is based on the use of long-term average atmospheric dilution factors. Under these conditions, it would be appropriate to average the emissions over a period of one year. In practice, however, the emission rate is not constant and a shorter averaging time is required in order to ensure that, in the unlikely event that a significant fraction of the DRL is discharged in a short period of time, the annual dose limit is not exceeded. An appropriate averaging time should be determined on a site-specific basis taking account of factors such as the magnitude of likely variations in the emission rate, the normal level of releases, and the probability of the various weather conditions. Experience has shown that for CANDU nuclear power stations where the releases are relatively constant at a small fraction of the annual DRL, an averaging time of one week is appropriate." In our estimation, an averaging time of one week is also appropriate for uranium mines and mills.

"The methods of this report are not meant to be applicable to the unlikely case where more than about 1% of the annual DRL is released in a matter of a few hours."

8.1.2 Surface Water Releases

"Aquatic dispersion is generally a slower process than atmospheric dispersion. For releases into rivers, the ratio of mean to peak dilution factors available at a given point downstream, after the initial mixing zone, is close to unity. Dispersion modelling for the Great Lakes (USNRC, 1977a) also suggests that the ratio of annual average to short-term dilution factors is rather small. This modelling was based on the observation that the discharge plume tends to hug the shoreline in one direction for several days, followed by relatively quick dispersion offshore and then development of a plume in the opposite direction.

The averaging time for calculating the DRL for aquatic releases is one month. This will provide reasonable assurance that, in the unlikely event that the monthly DRL amount is discharged in a few hours when dispersion conditions may be somewhat worse than average, the resulting maximum individual dose is unlikely to exceed the annual dose limit."

8.1.3 Groundwater Releases

Aquatic dispersion in groundwater is generally a slower process than in surface waters. Adsorption of radionuclides into geological materials and soils further reduces the rate of aquatic dispersion relative to that for conservative (non-adsorbing) salts. The time of transport in groundwater is generally slower than surface waters.

Alternative averaging times for calculating groundwater releases vary from one month to one year. Provided that the time of transport from point of release to groundwater to the critical group is more than one year, an averaging time of one year is recommended.

The time of transport from the point of release to groundwater to the point at which it impacts the critical group should also be compared to the operating life of the facility. For many geological materials, the time of transport of a conservative constituent through 100 m will be of the order of operating life of a facility (20 years). For radionuclides which adsorb to the geological materials, the time of transport may be longer than the operating life of the facility. The relative transport times can be evaluated using the models described in Chapter 7 and Appendix D.

For radionuclides whose transport time to the location of the critical group is longer than the operating life of the facility, the calculation of a DRL may not be appropriate because the concepts underlying the DRL are based on the assumptions that:

1. discharges can be managed by the operator, and
2. the impact of the radionuclide release is felt during the operational lifetime of the facility.

Such discharges will impact humans after operation of the facility has ceased, and hence the concept of the operator controlling annual operational dose may not apply. If the concept of an environmental committment to dose for future operations were to be included in the overall DRL concept, then the groundwater pathway would be of potentially major importance.

8.2 Additivity Effects from Different Radionuclides, Pathways and Sources

"Account should be taken of exposure from all radionuclides and release pathways (airborne and liquid) that contribute to any dose received by members of the critical group to ensure the appropriate dose limits are not exceeded. At a multi-facility site, total exposure from releases from different source facilities should also be considered. Dose limitation requirements will be met if during operation:

$$\sum_i \sum_j \sum_k \frac{R_{ijk}}{DRL_{ijk}} \leq 1 \qquad (8.1)$$

where R_{ijk} = Actual release of radionuclide i from effluent pathway j (air, surface water or groundwater) and source facility k;

DRL_{ijk} = Derived Release Limit for radionuclide i, effluent pathway j (air, surface water or groundwater) and source facility k.

"It is possible that some values of DRL_{ijk} will be based on the stochastic dose limit and others on the non-stochastic dose limit, depending on the radionuclide released (see Section 4.2). Nevertheless, application of Equation 8.1 will ensure that neither dose limit will be exceeded for a muitiplicity of different radionuclides released.

Equation 8.1 is expected to be conservative, particularly for a number of facilities on a large site, where the critical group for one nuclear facility may be different from that for another facility on the same site. In practice, somewhat modified expressions may be used to satisfy the intent of this Section."

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APPENDIX A

Default Values for Transfer Parameters

Note: This Appendix contains default values recommended for use in the absence of site-specific data. Default values given in this Appendix should be used only as a last resort; transfer parameters are often highly variable and the default values may not be representative of the specific pathways being assessed.

TABLE A1: TRANSFER PARAMETER P_{13} ATMOSPHERE TO VEGETATED SOIL

Radionuclide	P_{13} (m)	
	*	**
U-238+	1.4E7	4.5E7
U-234	1.4E7	4.5E7
Th-230	1.4E7	4.5E7
Ra-226	1.3E7	4.3E7
Pb-210	5.1E4	1.7E5
Po-210	2.5E6	8.3E6
Th-232	1.4E7	4.5E7
Ra-228	8.6E5	2.9E6
Th-228	2.5E5	8.4E5

These values are derived as described in Section 5.3 (page 5.12).

* Assuming a deposition velocity (Vg) of 0.003 $m \cdot s^{-1}$.

** Assuming a deposition velocity of 0.01 $m \cdot s^{-1}$.

+ These transfer parameter values also apply to U-NAT.

TABLE A2: TRANSFER PARAMETER P_{14}
ATMOSPHERE TO FORAGE, CROPS, BROWSE AND LICHEN

Radionuclide	P_{14} ($m^3 \cdot kg^{-1}$)			
	Forage	Vegetable	Browse	Lichen
U-238[+]	4.8E4	1.2E4	1.7E5	9.5E6
U-234	4.8E4	1.2E4	1.7E5	9.5E6
Th-230	4.8E4	1.2E4	1.7E5	9.5E6
Ra-226	4.8E4	1.2E4	1.7E5	9.5E6
Pb-210	4.6E4	1.1E4	1.6E5	8.9E5
Po-210	4.8E4	1.2E4	1.7E5	8.4E6
Th-232	4.8E4	1.2E4	1.7E5	9.5E6
Ra-228	4.8E4	1.2E4	1.7E5	6.6E6
Th-228	4.8E4	1.2E4	1.7E5	3.4E6

These values are derived as described in Section 5.4 (page 5.13).

[+] These transfer parameter values also apply to U-NAT.

TABLE A3: TRANSFER PARAMETER P_{34} PLANT UPTAKE FROM SOIL

	P_{34} ($m^2 \cdot kg^{-1}$)		
Element	Forage	Fruit and Vegetables	Browse
U	4.2E-5	2.1E-6	3.1E-4
Th	1.3E-5	2.1E-6	6.1E-4
Ra	8.3E-6	2.1E-5	6.3E-3
Pb	1.7E-5	4.2E-6	3.1E-3
Po	1.7E-5	8.3E-7	6.3E-4

These values are derived as described in Section 5.7 (page 5.17).

TABLE A4: TRANSFER PARAMETERS P_{45} AND P_{75} ($kg \cdot kg^{-1}$)
TRANSFER FROM VEGETATION TO ANIMAL TISSUE AND AQUATIC MACROPHYTES TO MOOSE

	P_{45}							P_{75}
Element	Milk	Egg	Beef	Pork	Chicken	Moose	Caribou	Moose
U	1E-3	2E-1	3E-1	2E-1	1.2E-1	3E-2	2E-3	1E-3
Th	5E-4	-	1E-3	-	-	5E-3	1E-3	2E-4
Ra	1E-2	2E-2	1E-2	3E-3	-	2E-2	1E-2	8E-4
Pb	1E-4	-	8E-3	-	-	2E-2	8E-3	8E-4
Po	5E-3	-	3E-1	-	-	3E-1	2E-1	1E-2

These values are derived as described in Section 5.8 (page 5.18).

TABLE A5: TRANSFER PARAMETER P_{15}
TRANSFER FROM ATMOSPHERE TO ANIMAL PRODUCE

	P_{15} ($m^3 \cdot kg^{-1}$)						
Element	Milk	Egg	Beef	Pork	Chicken	Moose	Caribou
U	4.7E-2	6.3E0	4.0E0	3.0E0	3.8E0	9.4E-1	3.6E-2
Th	3.4E-2	-	2.0E-2	-	-	2.1E-1	3.6E-2
Ra	2.5E-1	3.4E-1	7.2E-2	2.7E-2	-	3.2E-1	9.6E-2
Pb	6.2E-3	-	1.4E-1	-	-	7.8E-1	1.9E-1
Po	2.1E-1	-	3.6E0	-	-	7.7E0	3.3E0

These values are derived as described in Section 5.9 (page 5.19).

TABLE A6: TRANSFER PARAMETER P_{24}
CONTAMINATION OF VEGETATION BY SPRAY IRRIGATION WATER

Radionuclide	P_{24} ($L \cdot kg^{-1}$)
U-238+	1.2E0
U-234	1.2E0
Th-230	1.2E0
Ra-226	1.2E0
Pb-210	1.1E0
Po-210	1.2E0
Th-232	1.2E0
Ra-228	1.2E0
Th-228	1.2E0

These values are derived as described in Section 6.2 (page 6.5).

+ These transfer parameter values also apply to U-NAT.

TABLE A7: TRANSFER PARAMETER P_{23} CONTAMINATION OF SOIL BY IRRIGATION WATER

Radionuclide	P_{23} ($L \cdot m^{-2}$)
U-238[+]	9.1E4
U-234	9.1E4
Th-230	9.1E4
Ra-226	8.7E4
Pb-210	3.4E2
Po-210	1.7E4
Th-232	9.1E4
Ra-228	5.7E3
Th-228	1.7E3

These values are derived as described in Section 6.3 (page 6.6).

[+] This transfer parameter value also apply to U-NAT.

TABLE A8: TRANSFER PARMAETER P_{25} WATER INTAKE BY LAND ANIMALS

Element	P_{25} (L·kg^{-1})						
	Milk	Egg	Beef	Pork	Chicken	Moose	Caribou
U	8E-3	6E-1	1.5E0	4.2E-1	4E-1	3E-1	8E-3
Th	4E-3	-	5E-3	-	-	5E-2	6E-3
Ra	8E-2	6E-2	5E-2	7E-3	-	2E-1	5E-2
Pb	8E-4	-	4E-2	-	-	2E-1	4E-2
Po	2.4E0	-	1.5E0	-	-	3E0	8E-1

These values are derived as described in Section 6.4 (page 6.6).

TABLE A9: RADIONUCLIDE BIOACCUMULATION FACTORS FOR FRESHWATER BIOTA, P_{26}, P_{27} ($L \cdot kg^{-1}$)

Element	Fish	Macrophytes	
		Whole	Rice
U	2E1*/2E0**	4E3	2E1
Th	3E1	6E3	3E1
Ra	5E1	1E2	5E1
Pb	3E2	8E2	4E0
Po	5E1	8E3	4E1

* Value for low mineral content waters, total dissolved solids (TDS) concentration of <150 mg/L.

** Value for high mineral content waters, TDS ≥ 150 mg/L.

Fish data from Letourneau (1984); CSA (1984); Thompson et al. (1972), based on fresh weight.

Macrophyte data from Thompson et al. (1972), divided by 0.25 to convert to dry weight (whole macrophytes only); rice based on fresh weight.

TABLE A10: TRANSFER PARAMETERS P_{26}, P_{27} ($L \cdot kg^{-1}$)
RADIONUCLIDE BIOACCUMULATION FACTORS FOR MARINE BIOTA

Element	Fish Flesh	Marine Shellfish	Seaweed
U	1E-1	1E1	1E1
Th	1E3	1E3	1E3
Ra	1E2	1E2	1E3
Pb	3E2	1E2	1E3
Po	2E3	2E4	1E3

Data from Letourneau (1984) and Mitchell (1980).

TABLE A11: TRANSFER PARAMETER P'_{28} ($L \cdot kg^{-1}$ dry weight) RADIONUCLIDE DISTRIBUTION COEFFICIENT BETWEEN SEDIMENT AND WATER[3]

Element	Freshwater Sediment[1]	Marine Sediment[2]
U	2.5E4	5E2
Th	9E4	5.0E6
Ra	2E4	5.0E2
Pb	2E5	1.0E4
Po	7E4	1.0E4

[1] Freshwater values estimated from partition coefficients measured by Brunskill (1984) in a Canadian Shield Lake.

[2] Marine values provided by Mitchell (1980).

[3] Transfer coefficient of text (Equation 6.14) has units of L/kg wet weight while this table has units of L/kg dry weight. Values for P_{28} of text can be calculated by:

$$P_{28} = P'_{28}(\varepsilon\rho_w + (1-\varepsilon)\rho_s)/(\rho_s(1-\varepsilon))$$

where: ε is the porosity of sediment

ρ_w is the density of water (e.g., 1 gm/cc)

ρ_s is the density of the sediment solids (e.g., 1.2-1.3 g/cc for organic particles; 2.65 for quartz sand)

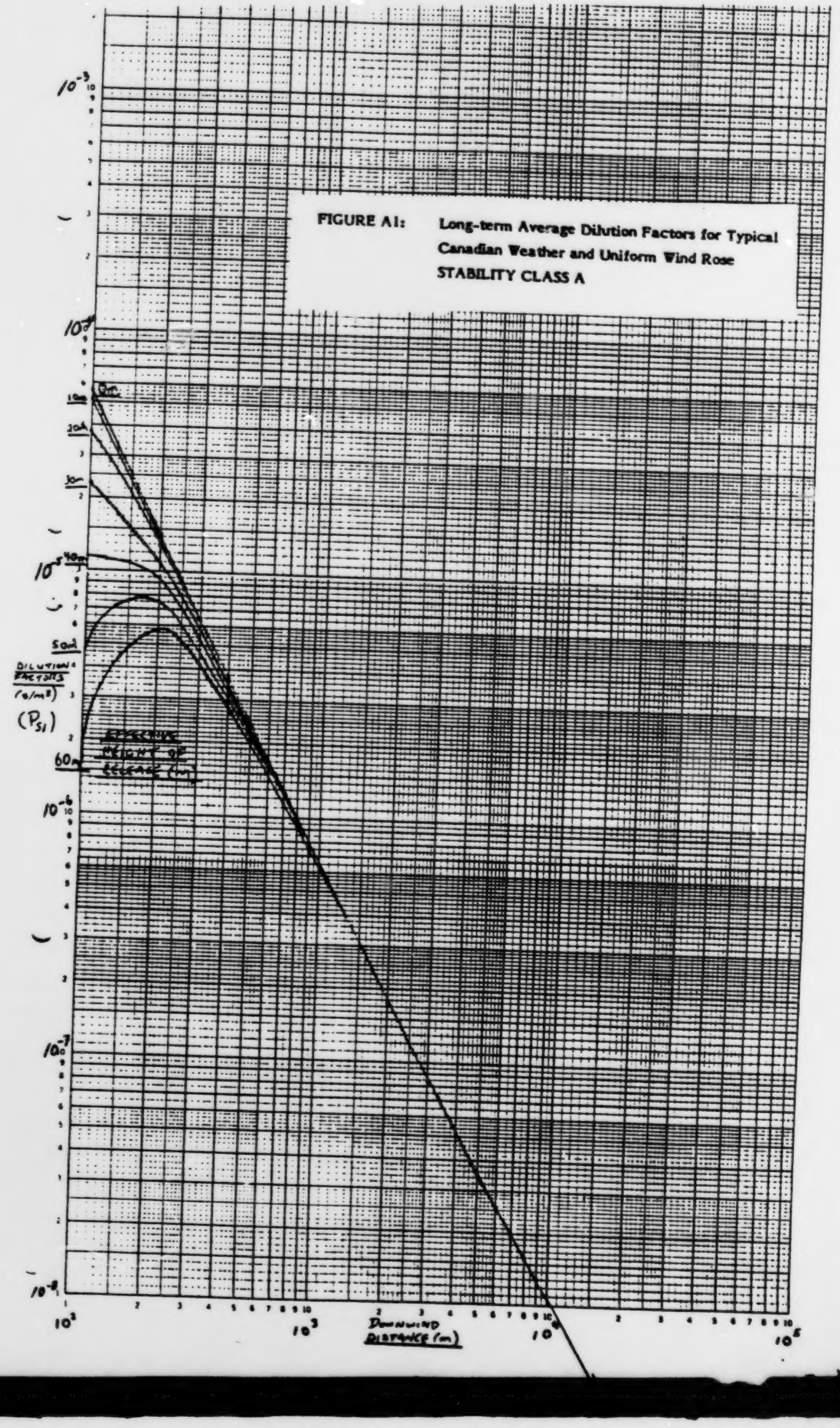

FIGURE A1: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose
STABILITY CLASS A

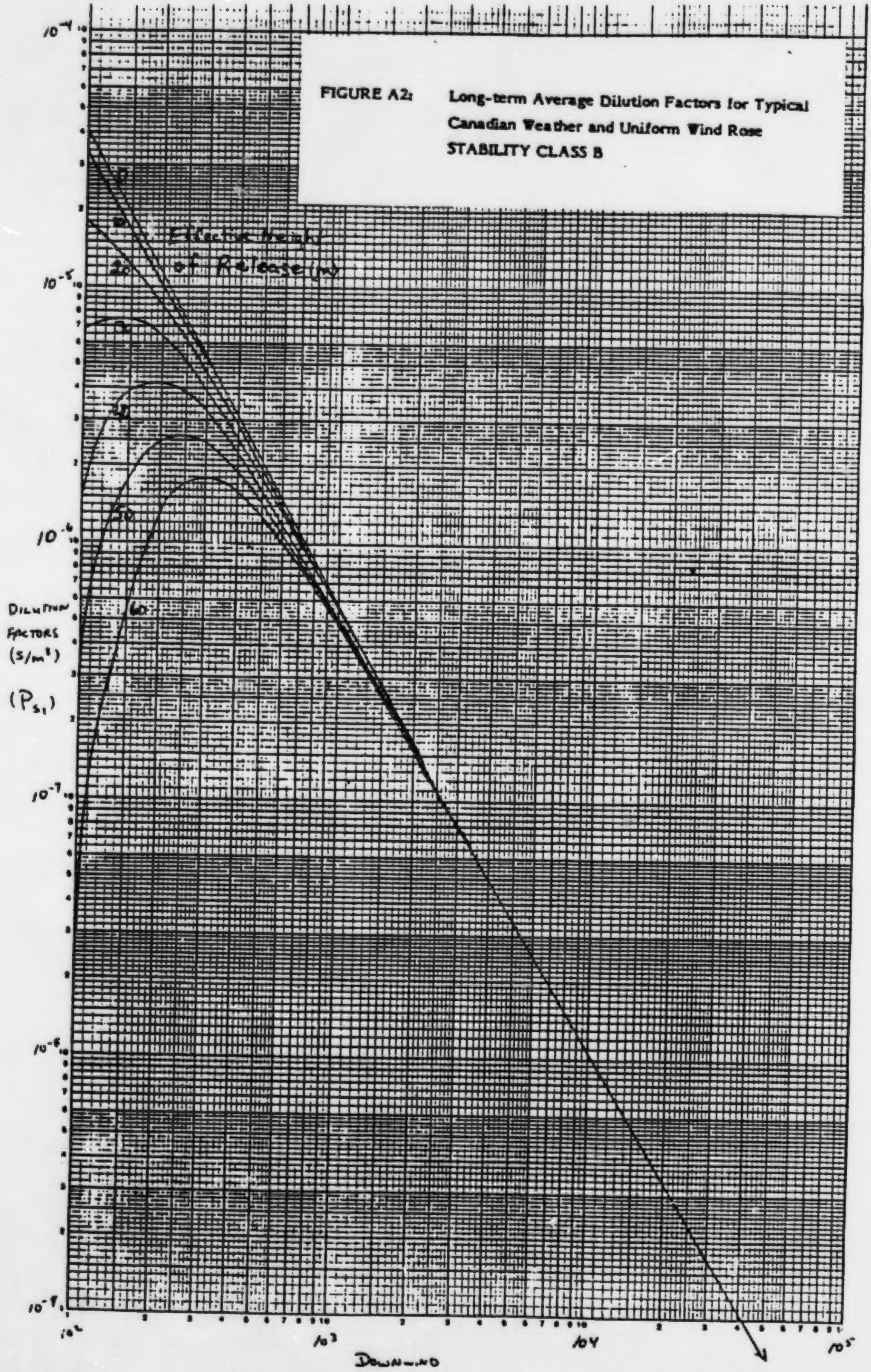

FIGURE A2: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose
STABILITY CLASS B

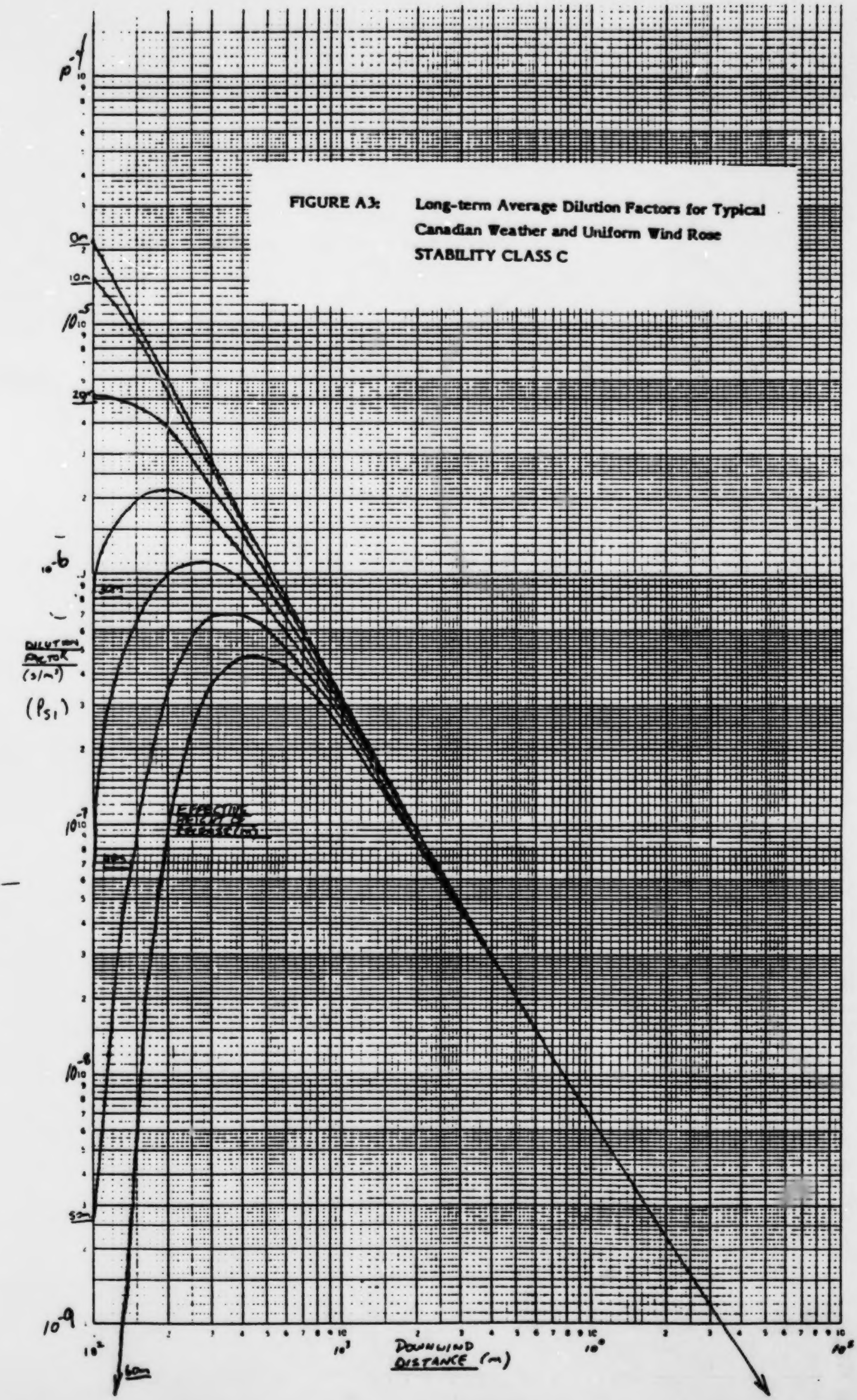

FIGURE A3: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose STABILITY CLASS C

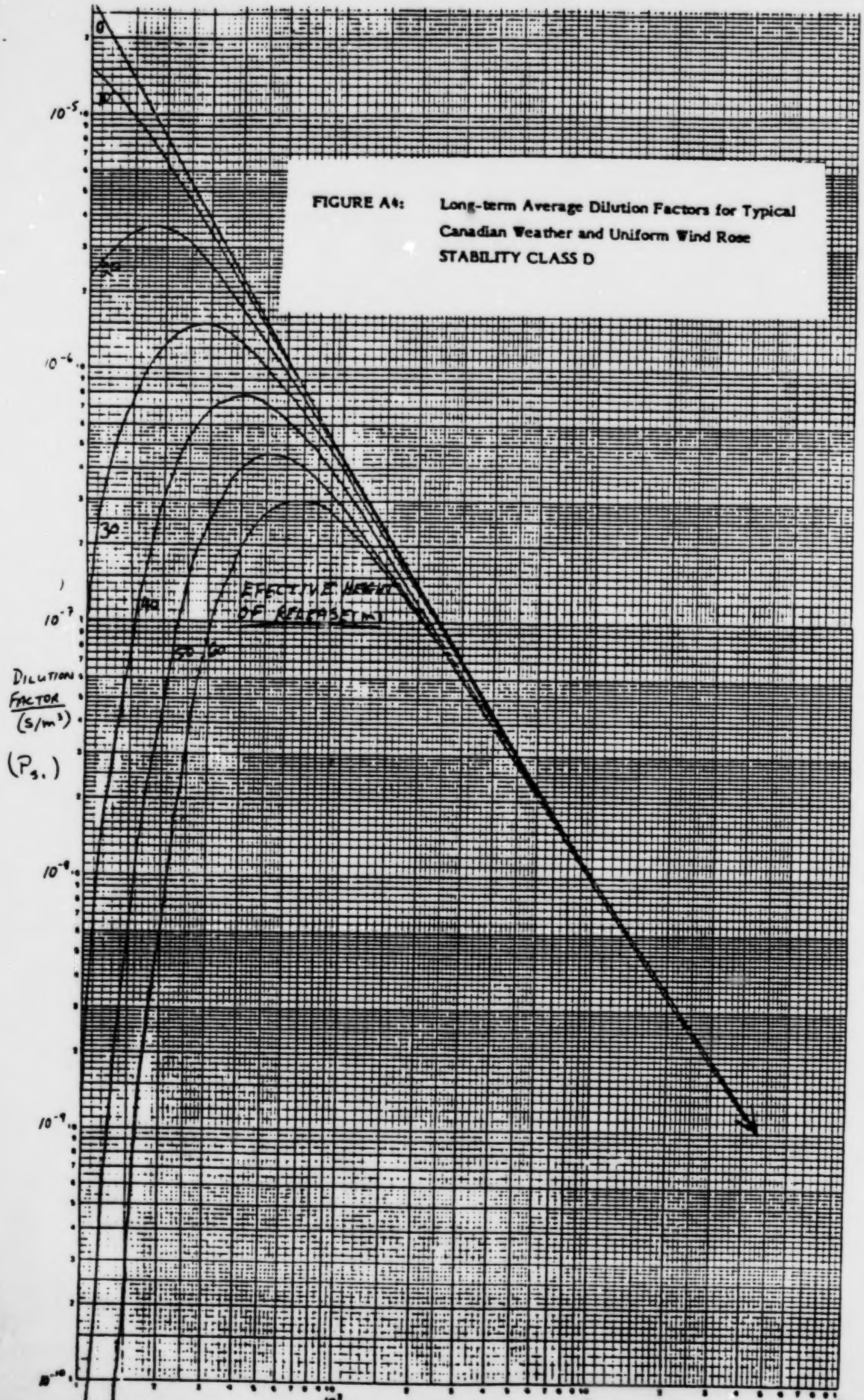

FIGURE A4: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose
STABILITY CLASS D

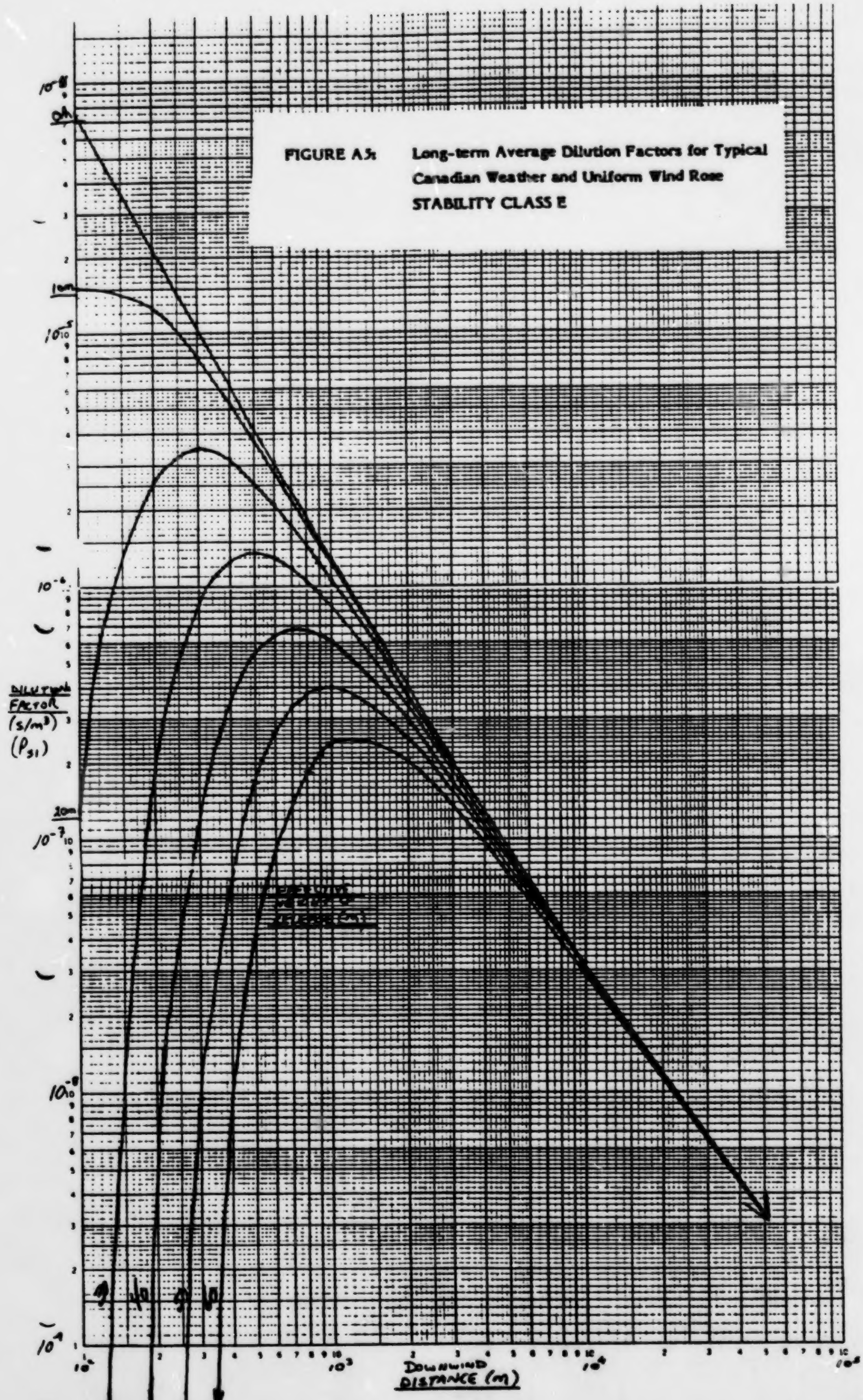

FIGURE A5: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose
STABILITY CLASS E

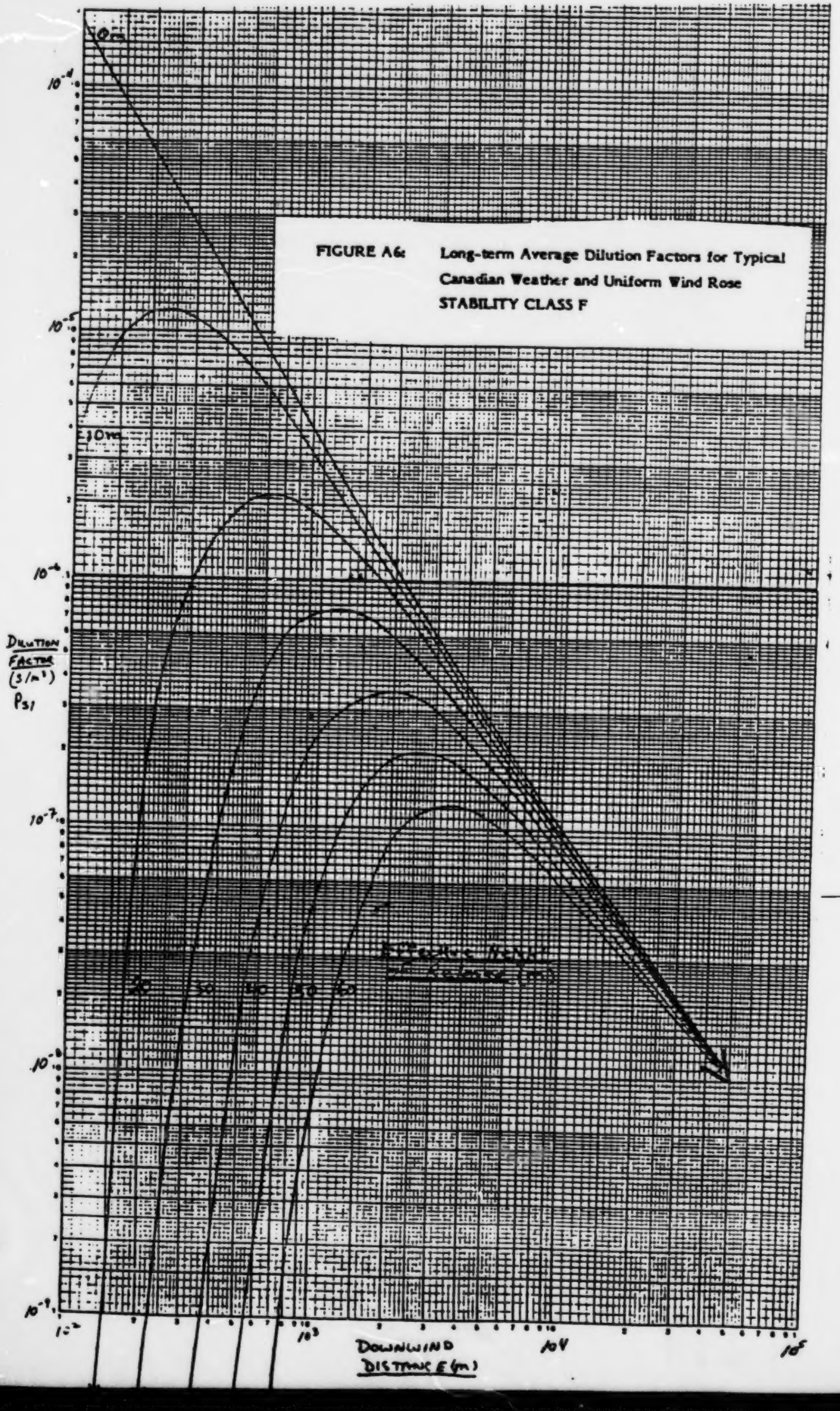

FIGURE A6: Long-term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose STABILITY CLASS F

APPENDIX B

Dose Conversion Factors

The most recent recommended dose conversion factors should always be used. This set is the most appropriate at the time of completion of this report.

TABLE B1: COMMITTED EFFECTIVE DOSE EQUIVALENT CONVERSION FACTORS (Sv/Bq)

Radionuclides	Inhalation Class+	Inhalation $(DCF)_i$***		Ingestion $(DCF)_f$***		
		Adult	Infant	f_1+	Adult	Infant
U-238	Y	3.8E-5	1.8E-4	0.05	7.5E-8*	1.3E-7*
U-234	Y	4.3E-5	2.1E-4	0.05	8.5E-8*	1.4E-7*
U-NAT++	Y	1.0E-6	4.9E-6	0.05	2.0E-9*	3.4E-9*
Th-230	Y	8.1E-5*	3.1E-4	0.03	2.1E-5*	5.5E-5*
Ra-226	W	2.6E-6	8.2E-6	0.2	3.2E-7*	1.2E-6*
Pb-210	D	3.5E-6*	1.5E-5*	0.2	1.6E-6*	6.9E-6*
Po-210	W	2.5E-6	6.9E-6	0.1	5.3E-7	8.0E-7
Th-232	Y	3.7E-4*	1.3E-3	0.03	1.1E-4*	1.8E-4*
Ra-228	W	1.3E-6	3.5E-6	0.2	2.6E-7*	5.3E-7*
Th-228	Y	1.1E-4	4.3E-4	0.03	1.5E-5*	9.8E-5*
Rn-222		**	**	-	-	-

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* For these radionuclides, pathways and age groups, the non-stochastic dose limit for an organ may be limiting. Therefore, a separate calculation using the organ dose conversion factors given in Table B2 should also be made.

** The dose associated with Rn-222 releases results from inhalation of the short-lived daughters. The dose conversion factor for inhalation of radon daughters is 1.3×10^4 $Sv \cdot a^{-1} \cdot J^{-1} \cdot m^3$ which applies to members of the public of all age groups (see Appendix F and Table F1). The calculation of the transfer parameter $P(i)_{19}$ for Rn-222 gas is given in Section 5.6.

*** From Johnson and Dunford (1983).

+ The inhalation class is an indication of the length of time the radioactive material is held in the lungs (D = relatively soluble; material remains in lungs for Days. W = slightly soluble; material remains in lungs for Weeks. Y = insoluble; material remains in lungs for Years). The f_1 value is the fraction of radioactive material reaching body fluids after entry into the gut. Both the inhalation class and the f_1 values depend on the chemical and physical form of the radionuclide. Values listed correspond to the maximum value of the Dose Conversion Factor for each radionuclide.

++ DCF's for U-NAT are in units of $Sv \cdot ug^{-1}$, and are calculated based on the specific activity relationships given in Section 4.3. U-NAT consists of 99.275 atom % U-238 (includes contributions from Th-234 and Pa-234 m), 0.0054 atom % of U-234, and 0.72 atom % of U-235 (includes contribution from Th-231).

TABLE B2: COMMITTED DOSE EQUIVALENT CONVERSION FACTORS (Sv/Bq) FOR RADIONUCLIDES AND ORGANS FOR WHICH THE NON-STOCHASTIC DOSE LIMIT MAY BE LIMITING

	Inhalation $(DCF)_i$**				Ingestion $(DCF)_f$**			
Radionuclide	Inhalation Class	Adult	Infant	Target Organ*	f_1[+]	Adult	Infant	Target Organ*
U-238	D	8.5E-6	1.6E-5	BS	0.05	9.9E-7	1.9E-6	BS
U-234	D	9.7E-6	1.9E-5	BS	0.05	1.1E-6	2.2E-6	BS
U-NAT[++]	D	2.3E-7	4.4E-7	BS	0.05	2.7E-8	5.2E-8	BS
Th-230	Y	9.6E-4	2.6E-3	BS	0.03	5.2E-4	1.4E-3	BS
Ra-226	W	6.3E-6	2.3E-5	BS	0.2	6.8E-6	2.6E-5	BS
Pb-210	D	4.6E-5	2.4E-4	BS	0.2	2.1E-5	1.1E-4	BS
Th-232	Y	5.8E-3	1.2E-2	BS	0.03	2.8E-3	4.6E-3	BS
Ra-228	W	5.1E-6	1.2E-5	BS	0.2	5.4E-6	1.1E-5	BS
Th-228	Y	1.6E-3	2.4E-4	BS	0.03	3.6E-4	2.4E-3	BS

* BS = Bone Surface.

** From Johnson and Dunford (1983 and pers. comm., 1985).

[+] See footnote under Table B1.

[++] See footnote under Table B1.

TABLE B3: DOSE CONVERSION FACTORS, $(DCF)_a$, FOR IMMERSION IN A CLOUD OF RADIOACTIVE MATERIAL ($Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^3$)

Radionuclides	Effective Dose*	Skin Dose**
U-238 Series		
U-238^{+}	2.67E-8	1.76E-6
U-234	2.06E-10	8.7E-10
U-NAT^{++}	4.64E-10	2.20E-8
Th-230	5.29E-10	1.09E-9
Ra-226	9.26E-9	1.47E-8
Rn-222^{+}	2.53E-6	5.08E-6
Pb-210^{+}	1.81E-9	7.18E-7
Po-210	1.19E-11	1.62E-11
Th-232 Series		
Th-232	2.52E-10	7.77E-10
Ra-228^{+}	1.31E-6	2.44E-6
Th-228^{+}	1.64E-8	2.39E-8

* From Kocher (1983), excluding the weighted skin dose. Values listed apply for adults; for infants, multiply by 1.5.

** From Kocher (1981). Sum of the skin gamma dose factors and the electron dose factors at a depth of 70 um below the body surface. Values listed apply for adults and infants.

$^{+}$ Includes contribution from daughters closely associated with the parent (see Section 4.3). Specifically:

U-238^{+} = U-238 + Th-234 + Pa-234m
Rn-222^{+} = Rn-222 + Po-218 + Pb-214 + Bi-214 + Po-214
Pb-210^{+} = Pb-210 + Bi-210
Ra-228^{+} = Ra-228 + Ac-228
Th-228^{+} = Th-228 + Ra-224

The daughters are assumed to be in secular equilibrium with the parents.

$^{++}$ See footnote under Table B1.

TABLE B4: DOSE CONVERSION FACTORS, $(DCF)_g$, FOR EXPOSURE 1 M ABOVE UNIFORMLY CONTAMINATED PLANE SURFACE ($Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^2$)

Radionuclides	$DCF_{(g)}$ Effective Dose*	Skin Dose**
U-238 Series		
U-238⁺	6.0E-10	3.0E-7
U-234	2.18E-11	1.72E-10
U-NAT⁺⁺	1.07E-11	3.73E-9
Th-230	2.45E-11	1.4E-10
Ra-226⁺	4.55E-8	2.91E-7
Pb-210⁺	8.12E-11	1.07E-7
Po-210	2.33E-13	3.15E-13
Th-232 Series		
Th-232⁺	6.38E-8	4.19E-7
Ra-228⁺	6.38E-8	4.19E-7
Th-228⁺	3.91E-8	2.84E-7

* From Kocher (1983), excluding the weighted skin dose. Values listed apply for adults; for infants, multiply by 1.5.

** From Kocher (1981). Sum of the skin gamma dose factors and the electron dose factors at a depth of 70 um below the body surface. Values listed apply for adults and infants.

+ Includes contribution from daughters closely associated with the parent (see Section 4.3). Specifically:

$U\text{-}238^+$ = U-238 + Th-234 + Pa-234m
$Ra\text{-}226^+$ = Ra-226 + Rn-222 + Po-218 + Pb-214 + Bi-214 + Po-214
$Pb\text{-}210^+$ = Pb-210 + Bi-210 + Po-210
$Th\text{-}232^+$ = Th-232 + all its decay products
$Ra\text{-}228^+$ = Ra-228 + all its decay products
$Th\text{-}228^+$ = Th-228 + all its decay products

The daughters are assumed to be in secular equilibrium with the parents.

++ See footnote under Table B1.

TABLE B5: DOSE CONVERSION FACTOR, $(DCF)_w$, FOR IMMERSION IN CONTAMINATED WATER ($Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$)

Radionuclides	DCF$_{(w)}$ Effective Dose*	Skin Dose**
U-238 Series		
U-238$^+$	6.0E-8	1.92E-6
U-234	5.04E-10	2.15E-9
U-NAT^{++}	1.04E-9	2.42E-8
Th-230	1.25E-9	2.66E-9
Ra-226	2.06E-8	2.95E-8
Rn-222$^+$	5.48E-6	9.12E-6
Pb-210$^+$	4.55E-9	7.66E-7
Po-210	2.59E-11	3.5E-11
Th-232 Series		
Th-232	6.1E-10	1.89E-9
Ra-228$^+$	2.82E-6	4.53E-6
Th-228$^+$	3.62E-8	4.60E-8

* From Kocher (1983), excluding the weighted skin dose. Values listed apply for adults; for infants, multiply by 1.5.

** From Kocher (1981). Sum of the skin gamma dose factors and the electron dose factors at a depth of 70 um below the body surface. Values listed apply for adults and infants.

+ Includes contribution from daughters closely associated with the parent (see Section 4.3). Specifically:

U-238$^+$ = U-238 + Th-234 + Pa-234m
Rn-222$^+$ = Rn-222 + Po-218 + Pb-214 + Bi-214 + Po-214
Pb-210$^+$ = Pb-210 + Bi-210
Ra-228$^+$ = Ra-228 + Ac-228
Th-228$^+$ = Th-228 + Ra-224

The daughters are assumed to be in secular equilibrium with the parents.

++ See footnote under Table B1.

APPENDIX C

Default Data for Calculation of Transfer Parameters

TABLE C1: DEFAULT STABILITY CATEGORY FREQUENCY DISTRIBUTION FOR TYPICAL CANADIAN WEATHER AND AVERAGE VERTICAL TEMPERATURE GRADIENT

Stability Category	Frequency* (%)	Mean Wind Speed $\bar{u}\ (m \cdot s^{-1})$	T/ Z** (°C/100 m)
A	1	1	≤ -1.9
B	6	2	$-1.9 < \Delta T/\Delta Z \leq -1.7$
C	10	5	$-1.7 < \Delta T/\Delta Z \leq -1.5$
D	56	5	$-1.5 < \Delta T/\Delta Z \leq 0.5$
E	10	3	$0.5 < \Delta T/\Delta Z \leq 1.5$
F	17	2	$1.5 < \Delta T/\Delta Z \leq 4.0$

* From CSA (1984).

** From USAEC (1972).

TABLE C2: EQUATIONS FOR CALCULATING PLUME DISPERSION PARAMETERS (Hosker, 1974)

$$\sigma_{zi}(x) = g_i(x) \cdot F(Z_0;x)$$

where

$$g_i(x) = a_1 x^{b_1}(1 + a_2 x^{b_2})^{-1} \; ;$$

and

$$F(Z_0;x) = \begin{cases} \ln\left[c_1 x^{d_1}\left(1 + \dfrac{1}{c_2 x^{d_2}}\right)\right]; & Z_0 > 10 \text{ cm} \\ \ln\left[c_1 x^{d_1}\left(\dfrac{1}{1 + c_2 x^{d_2}}\right)\right]; & Z_0 < 10 \text{ cm} \; . \end{cases}$$

The parameter Z_o is the surface roughness length, and some examples are given in Table C5 (Smith, 1972; Pasquill, 1974). For rural conditions, a value of $Z_o = 40$ cm is appropriate. Values of the other parameters are given in Tables C3 and C4.

TABLE C3: COEFFICIENTS OF THE FUNCTION $g_i(x)$, USED IN CALCULATING THE VERTICAL DISPERSION COEFFICIENT $\sigma_{zi}(x)$, FOR THE VARIOUS STABILITY CATEGORIES (x is given in metres)

Stability Category i	a_1	b_1	a_2	b_2
A	0.112	1.06	5.38×10^{-4}	0.815
B	0.130	0.950	6.52×10^{-4}	0.750
C	0.112	0.920	9.05×10^{-4}	0.718
D	0.098	0.889	1.35×10^{-3}	0.688
E	0.0609	0.895	1.96×10^{-3}	0.684
F	0.0638	0.783	1.36×10^{-3}	0.672

From CSA (1984)

TABLE C4: COEFFICIENTS OF THE FUNCTION $F(Z_0;x)$ USED IN CALCULATING $\sigma_{zi}(x)$ FOR THE VARIOUS ROUGHNESS LENGTHS (x is given in metres)

Roughness Length Z_0 (cm)	c_1	d_1	c_2	d_2
1	1.56	0.0480	6.25×10^{-4}	0.45
4	2.02	0.0269	7.76×10^{-4}	0.37
10	e[(a)]	0	0	0
40	5.16	-0.098	18.6	-0.225
100	7.37	-0.0957	4.29×10^{3}	-0.60
400	11.7	-0.128	4.59×10^{4}	-0.78

(a) e = base of natural logarithms = 2.71828.

From CSA (1984)

TABLE C5: TYPICAL SURFACES CORRESPONDING TO VARIOUS ROUGHNESS LENGTHS

Roughness Length Z_0 (cm)	Examples of Surfaces
1	Lawn grass, bodies of water
4	Ploughed land
10	Open grassland
40	Rural areas with mixed farming, woods, small villages
100	Cities or forests
400	Cities with tall buildings

From CSA (1984)

TABLE C6: RADIONUCLIDES FOR WHICH SKIN DOSE TO EFFECTIVE DOSE RATIO EXCEEDS 10

	Skin Dose/Effective Dose*		
Radionuclides	Air Immersion	Ground Exposure	Water Immersion
U-238$^+$	66	500	32
U-NAT	47	349	23
Pb-210$^+$	397	1,320	168

*From Tables B3, B4 and B5 (Appendix B).

TABLE C7: RECOMMENDED VALUES OF DEPOSITION VELOCITIES OF AIRBORNE PARTICULATES

Particle Mean Diameter (um)	Diameter Range (um)	Density ($g \cdot cm^{-3}$)	Deposition Velocity ($m \cdot s^{-1}$)
<1	-	-	0.003
1	-	2.4	0.01
1	-	8.9	0.01
5	1 to 10	2.4	0.01
35	10 to 80	2.4	0.088

Notes:

a) Value for sub-micron particles taken from CSA N288.1 Standard.

b) All other values taken from USNRC (1980). A particle density of 8.9 $g \cdot cm^{-3}$ applies to yellowcake dust.

TABLE C8: VALUES OF VEGETATION YIELD

Vegetation	$Y(Kg \cdot m^{-2})$
Forage grass	0.28 (dry weight)[1]
Above-ground leafy vegetables	1.9 (fresh weight)[1]
Non-leafy vegetables	0.57 (fresh weight)[1]
Below-ground vegetables	2.6 (fresh weight)[1]
Fruits	0.31 (fresh weight)[1]
Browse	0.1 (dry weight)[2]
Lichen	0.18 (dry weight)[3]

It will frequently be the case that the food ingestion pathway to man will not be based on a single crop, but on a mixture of several (e.g., a backyard vegetable garden). For such a case, a single value of the yield, based on a weighting of categories above according to the consumption rates in Table C15, is about 1.4 $kg \cdot m^{-2}$ (fresh weight).

[1] CSA (1984)

[2] IEC BEAK (1984)

[3] Miller (1976)

TABLE C9: CONCENTRATION FACTORS B_v SOIL-VEGETATION[1]

Element	Forage Grass	Vegetables	Browse
U	1E-2	5E-4	5E-2
Th	3E-3	5E-4	1E-1
Ra	2E-3	5E-3	1E0
Pb	4E-3	1E-3	5E-1
Po	4E-3	2E-4	1E-1

Forage grass and browse parameters are per unit dry weight; vegetables are per unit fresh weight.

[1] From data provided by Letourneau (1984).

TABLE C10: FOOD CONSUMPTION RATES (Q_f) FOR ANIMALS

Animal	Daily Food Consumption, Q_f ($kg \cdot day^{-1}$, dry weight)
Dairy Cows	10
Beef Cattle	10
Chickens	0.1
Pigs	3
Caribou	2.8
Moose	4.8 (0.5*)

Data for caribou taken from Garner (1972) and moose from Velovsky et al. (1975). All other data taken from N288.1 (CSA, 1984).

* A biomass of 0.5 $kg \cdot day^{-1}$ of aquatic macrophytes is also assumed to be eaten by moose; based on an average daily consumption of 2 kg (estimated from Velovsky et al., 1975) and an assumed dry weight content of 25%.

TABLE C11: FRACTION (F) OF DAILY INTAKE BY INGESTION APPEARING IN 1 KG OF ANIMAL TISSUE

Radionuclide	Milk	Beef	Pork	Eggs	Chicken	Moose	Caribou
U-238[+]	1E-4	3E-2	6E-2	2E0	1.2E0	6E-3	7E-4
U-234	1E-4	3E-2	6E-2	2E0	1.2E0	6E-3	7E-4
Th-230	5E-5	1E-4	-	-	-	1E-3	5E-4
Ra-226	1E-3	1E-3	1E-3	2E-1	-	4E-3	4E-3
Pb-210	1E-5	8E-4	-	-	-	4E-3	3E-3
Po-210	5E-4	3E-2	-	-	-	6E-2	7E-2
Th-232	5E-5	1E-4	-	-	-	1E-3	5E-4
Ra-228	1E-3	1E-3	-	2E-1	-	4E-3	4E-3
Th-228	5E-5	1E-4	-	-	-	1E-3	5E-4

From data provided by Letourneau (1984). F values for moose and caribou calculated by dividing concentration factors (Letourneau, 1984) by Q_f values provided in Table D11.

[+] These transfer parameter values also apply to U-NAT.

TABLE C12: BREATHING RATES[1] FOR ANIMALS

Animal	Mass (kg)	Breathing Rate (I_a ($m^3 \cdot day^{-1}$)
Dairy Cow	600	200
Beef Cattle	400	85
Pig	100	30
Laying Hen	2	2
Moose	400	90
Caribou	95	30

[1] Rates for domestic livestock provided by Dr. H. Bayley, Department of Nutrition, University of Guelph (pers. comm., 1985). Rates for moose and caribou estimated using the method developed by Ontario Hydro (1983).

TABLE C13: FRACTION (F') OF DAILY INTAKE BY INHALATION APPEARING IN 1 KG OF ANIMAL PRODUCE (day·kg^{-1})

Element	Milk	Egg	Beef	Chicken	Pork	Caribou	Moose
U	1.7E-4	3.3E0	5.0E-2	2.0E0	1.0E-1	1.2E-3	1.1E-2
Th	1.2E-4	-	2.5E-4	-	-	1.2E-3	2.5E-3
Ra	9.0E-4	1.8E-1	9.0E-4	-	9.0E-4	3.2E-3	3.8E-3
Pb	2.2E-5	-	1.8E-3	-	-	6.4E-3	9.2E-3
Po	7.5E-4	-	4.5E-2	-	-	1.1E-1	9.0E-2

These values are derived as described in Section 5.9 (page 5.19).

TABLE C14: FOOD CONSUMPTION RATES ($kg \cdot a^{-1}$)

Food Category	Average Consumption[1]		Maximum Consumption	
	Infant	Adult	Infant	Adult
Leafy vegetables	10	14	12	30
Above-ground vegetables and fruits	50	110	60	250
Root vegetables	24	79	38	180
Cereal	12	74	18	120
Beef and Pork	24	71	39	80
Poultry	10	16	12	20
Milk	220	120	370	310
Milk Products	-	54	-	140
Eggs	5	14	7	28
Moose[2]	-	-	4	40
Caribou[2]	-	-	2	20
Aquatic Foods[3]				
Fish (freshwater)[4]	0.6	5.5	4.5	45
Wild Rice[4]	-	-	3.7	37
Fish (marine)[5]	1	10	3	30
Crustacean Flesh[5]	0.2	1.5	0.5	5
Mollusc Flesh[5]	0.2	1.5	0.5	5
Edible Seaweed (Dulse)[5]	0.3	3	1	10

---

[1] Average consumption rates as in CSA (1984), except for caribou, moose and aquatic foods.

[2] Moose and caribou consumption are assumed to be 50% and 25%, respectively, of beef and pork consumption.

[3] Intake rates of aquatic foods for infants are one-tenth those for adults. Intake rates for fish, crustacea and molluscs are based on fresh (wet) weight. Rate for seaweed is based on dry weight.

[4] Freshwater fish and wild rice consumption rates are our estimates (see Section 6.10).

[5] Consumption rates for seafoods are based on New Brunswick survey data (NBEPC, 1976).

TABLE C15: WATER INTAKE RATES FOR ANIMALS (Q_w)

Animal	Q_w ($L \cdot day^{-1}$)
Dairy Cows	80
Beef Cattle	50
Chickens	0.3
Pigs	7
Moose	50
Caribou	12

Livestock values from CSA (1984). Moose value from Velovsky et al. (1975). Caribou are assumed to drink at the same rate per kg body mass as moose.

TABLE C16: SHORE-WIDTH FACTORS FOR SHORELINE EXPOSURE

Exposure Location	Shore-Width Factor
River Shoreline	0.2
Lake Shore	0.3
Nominal Ocean Site	0.5
Tidal Basin	1

From U.S. NRC (1977a).

APPENDIX D

Models for Evaluation of Transport in Aquatic Systems

D.1 Basic Equations for One-Dimensional Transport

There are three basic transport models which may be used to describe the mixing characteristics of one-dimensional environmental systems: plug flow reactor model (PFR), continuously-stirred tank reactor model (CSTR; also termed "well-mixed" in limnological applications) and dispersed-plug flow reactor model (DPFR). The terminology is derived from chemical engineering. The PFR may be used to describe rivers, channels and groundwater flow through porous media (e.g., sands). The CSTR may be used to describe lakes, lagoons or near circular tanks. The DPFR may be used to describe mixing in estuaries, groundwater, or consolidating lake sediments, flow through clays or highly fractured media, and in rectangular tanks such as flocculation tanks, settling basins and aeration tanks in treatment systems. The transport models presented here are examples only; the application of any model for a specific situation must be justified.

The basic equation describing the one-dimensional (1-D) variation along direction x over time (t) of the concentration C of any material is

$$\frac{\delta C}{\delta t} = D \frac{\delta^2 C}{\delta x^2} - v \frac{\delta C}{\delta x} + \sum_i r_i \qquad \text{(D1)}$$

Here D is the longitudinal diffusivity coefficient, v the water velocity, and r the reaction rate describing all possible production rates or consumption rates of C by biochemical sources and sinks, physical fluxes such as volatilization, transport across the sediment-water interface, settling and radioactive decay. This equation neglects the effects of adsorption.

For typical pollutants of concern which undergo decay reactions:

$$r_i = kC^n \qquad \text{(D2)}$$

where k is the reaction constant and n is the order of the reaction. One of the most general phenomena observed is first order decay; examples include radioactive decay (e.g., Ra-226, with $t_{1/2} = 1620$ yr). For example, for a transport model describing Ra-226 which undergoes radioactive decay but no other removal reaction, $i = 1$; $k = \lambda_r$; $\lambda_r = 1.36 \times 10^{-11}\ s^{-1}$; and $n = 1$.

Equation DI conceptually describes all three mixing models. If diffusion is molecular and v is large (e.g., such that $vL >> D$ where L is the length scale of concern), $D \sim O$ and Equation DI becomes:

$$\frac{\delta C}{\delta t} = -v \frac{\delta C}{\delta x} + \sum_{i=1} r_i \tag{D3}$$

This is the basic governing equation of the PFR. On the other hand, if D approaches infinity, well mixed conditions result. Equations may be evaluated by allowing D to approach infinity, but as this occurs, $\delta C/\delta x$ approaches 0. This complication may be resolved by finding specific solutions to Equation DI or by deriving the equation for well mixed conditions. The new derivation gives:

$$\frac{\delta C}{\delta t} = \frac{C_o}{\tau_w} - \frac{C}{\tau_w} + \sum_{i=1} r_i \tag{D4}$$

where τ_w is the hydraulic detention time and C_o the average concentration of the material C entering the water body in all inflows.

Transport models for groundwater application are conceptually similar to those for estuaries. The major difference is the path taken by particles during transport. For dissolved constituents in saturated, homogeneous, isotropic materials under uniform flow, the advection - dispersion for "groundwater" regimes is presented by Freeze and Cherry (1979) as

$$\frac{\delta C}{\delta t} = \frac{D_l \, \delta^2 C}{\delta L^2} - \bar{v}_l \frac{\delta C}{\delta L} + \sum_{i=1} r_i \tag{D5}$$

Here, "L" is the curvilinear coordinate direction along the flow line, D_l is the coefficient of hydrodynamic dispersion in the longitudinal direction (along the flow path) and $\bar{v}_l$ is the average linear groundwater velocity. The equation is similar to Equation DI, but "L" represents a circuitous path (a path line) around the sands, gravel etc., particles which make up the parent porous media, as opposed to the 'x' coordinate described by Equation DI.

D.2 Mixing Characteristics of Transport Models

It is instructive to examine mixing characteristics of conservative and non-conservative materials in these three mixing models (PFR, DPFR, CSTR) before considering their quantitative descriptions. A conservative material is one having r=0. Two types of input to the reactors aid in differentiating their mixing characteristics - a step function and an impulse function. If a material is injected into a water body, it can be injected all at once (an impulse) or continuously (a step) such that the total mass injected continuously over a detention time is equal to the total mass injected during the impulse. The response curve observed in the outflow from the water body reactor, when normalized, is called an F curve (for the step input) and a C curve (for the impulse input). Both are probability distribution curves - the F curve is a cumulative distribution whereas the C curve is its derivative. Both can be used to determine the fraction of the discharge stream which has water elements having a particular residence time between two time periods. Typical relationships obtained for such curves strictly hold only when no back-mixing occurs between the inflow and outflow, but they form a useful approximation for determining mixing behaviour. For special problems caused back-mixing, the reader might consult Levenspiel (1962).

The F and C curves are shown in Figure Dl for the three reactors. For normalization purposes, the concentration of the step function is used for the F curve while the mass of the impulse function divided by the reactor volume is used for the C curve. The graphs show that no response is observed in the outflow from the PFR until one detention time whereas an immediate response is observed in the well mixed system. The DPFR shows a response intermediate between that of the PFR and CSTR. Hence the PFR and CSTR represent limiting cases. Because one rarely finds these assumptions fully met in nature, their utility depends partly on the degree of intermediate mixing which occurs and the degree of non-ideality that the modeller can tolerate without seriously compromising his objectives. Decisions on this point can be made by examining various time scales (Snodgrass, 1983a, b), e.g., for a PFR the ratio of vL/D; for a CSTR, the hydraulic or material detention time compared to the time to mix horizontally across the CSTR or the time scale of changes in inputs.

It is also instructive to consider the characteristics of conservative materials along the length of the reactor; this permits a fuller understanding of the behaviour in the

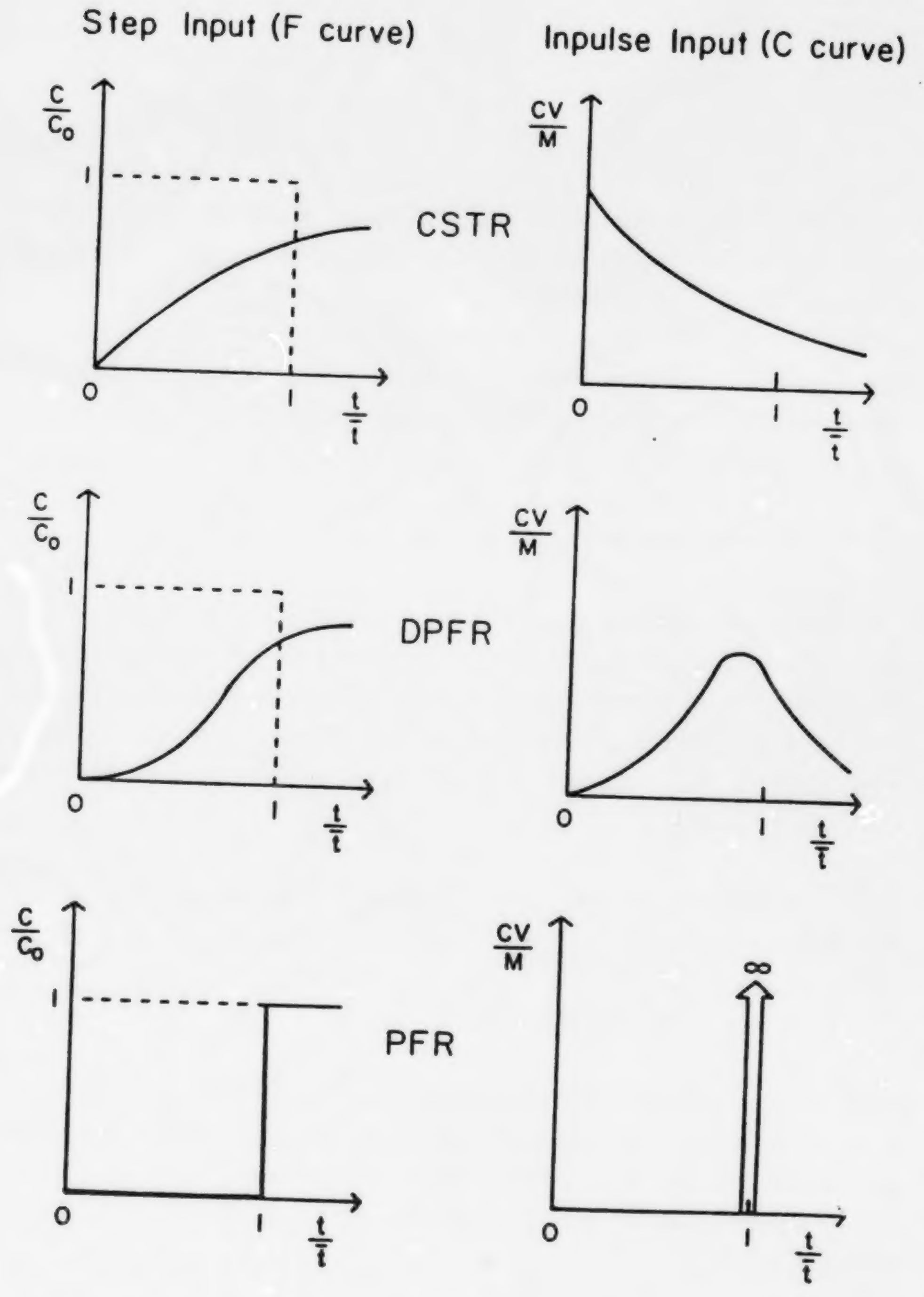

FIGURE D1: Normalized distribution curves resulting from step and impulse inputs for mixing models.

outflow. Figure D2 provides a few examples. The response of the CSTR is not shown because the response along the length of the tank is exactly the same as that of the outflow. For the PFR, the profile at an intermediate time and at $t > 1$ are given. For the DPFR, longitudinal profiles are given for $t < 0$; $t = 0$; $0 \leq t \leq 1$; $t > 1$ and $t \gg 1$; the relevant profiles for the PFR are also sketched for comparison. It shows that the effect of diffusion is to increase transport ahead of the "PFR front" and to dilute the concentration of material behind the "PFR front". It also shows that some material diffuses upstream of the injection point.

D.3 Mathematical Solutions for 1-D Transport Equations

The solutions for the PFR and CSTR equations are given in Table D1 for 0, 1st, 2nd and nth order kinetics. The results for first order kinetics are most directly applicable to the evaluation of radionuclide transport if no removal mechanism other than radioactive decay occurs. Then $k = \lambda_r$, where λ_r is the radioactive decay rate constant. If other mechanisms remove radionuclides from the water column, then one must evaluate the applicable kinetics.

Other reactions which may be of significance include settling of adsorbed forms, diffusion into sediments, and volatilization. The equation:

$$k = \lambda_r + g K_D M/(\bar{z} \lambda) \tag{D6}$$

describes the combined effects of decay (λ_r) and settling (gK_DM). Here, g is the settling velocity of particulate materials of concentration M, K_D is the linear distribution coefficient, $\bar{z}$ is the mean depth of the water body and ϵ is the porosity. For the dilute solids concentrations of open lake and river waters, the effects of ϵ are typically ignored because $\epsilon \sim 1$.

If diffusion is significant, a combined model which describes decay, settling and diffusion in a lake is:

$$\frac{\delta C_T}{\delta_t} = \frac{C_o}{\bar{t}_w} - \left[\frac{1}{\bar{t}_w} + \lambda_r + \frac{gK_D M}{\bar{z}} \right] C_T - \frac{k_s}{\bar{z}} \left[(1-f) \frac{C_T}{\epsilon} - \frac{(1-f_s)\, C_{TS}}{\epsilon_s} \right] \tag{D7}$$

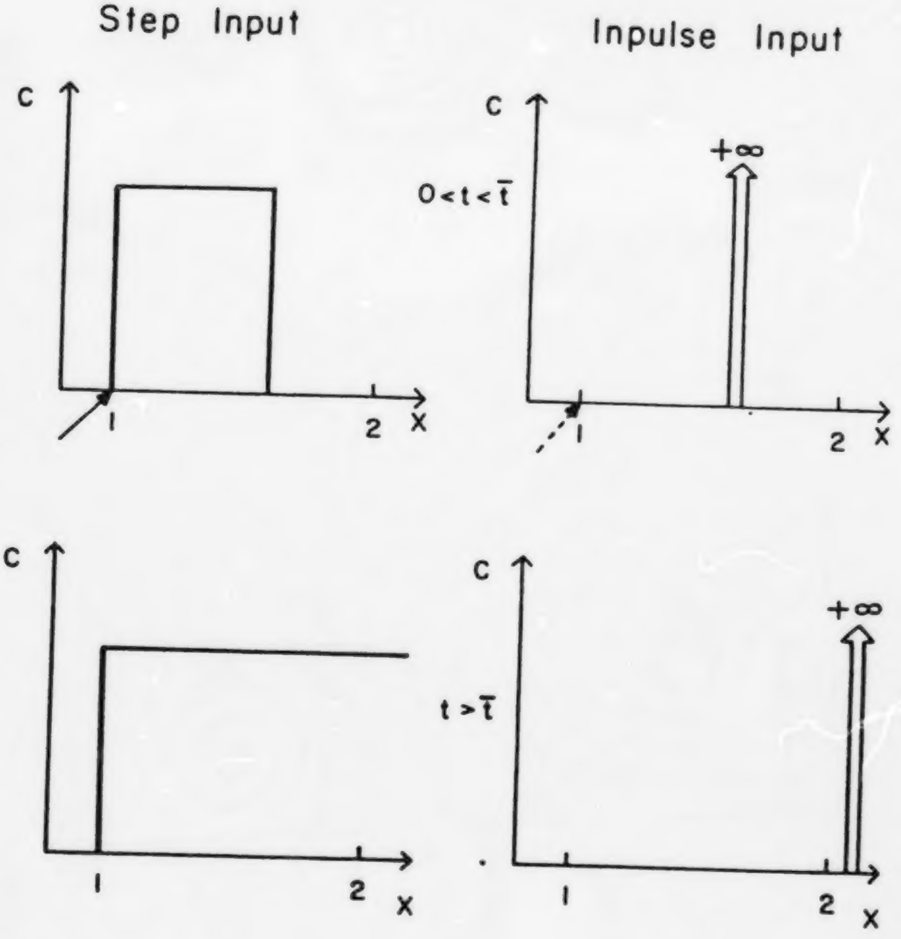
Step Input
Inpulse Input
C
C
0<t<t̄
+∞
1
2
X
1
2
X
C
C
t>t̄
+∞
1
2
X
1
2
X

FIGURE D2: a. Tracer curves along the length of PFR at various times.

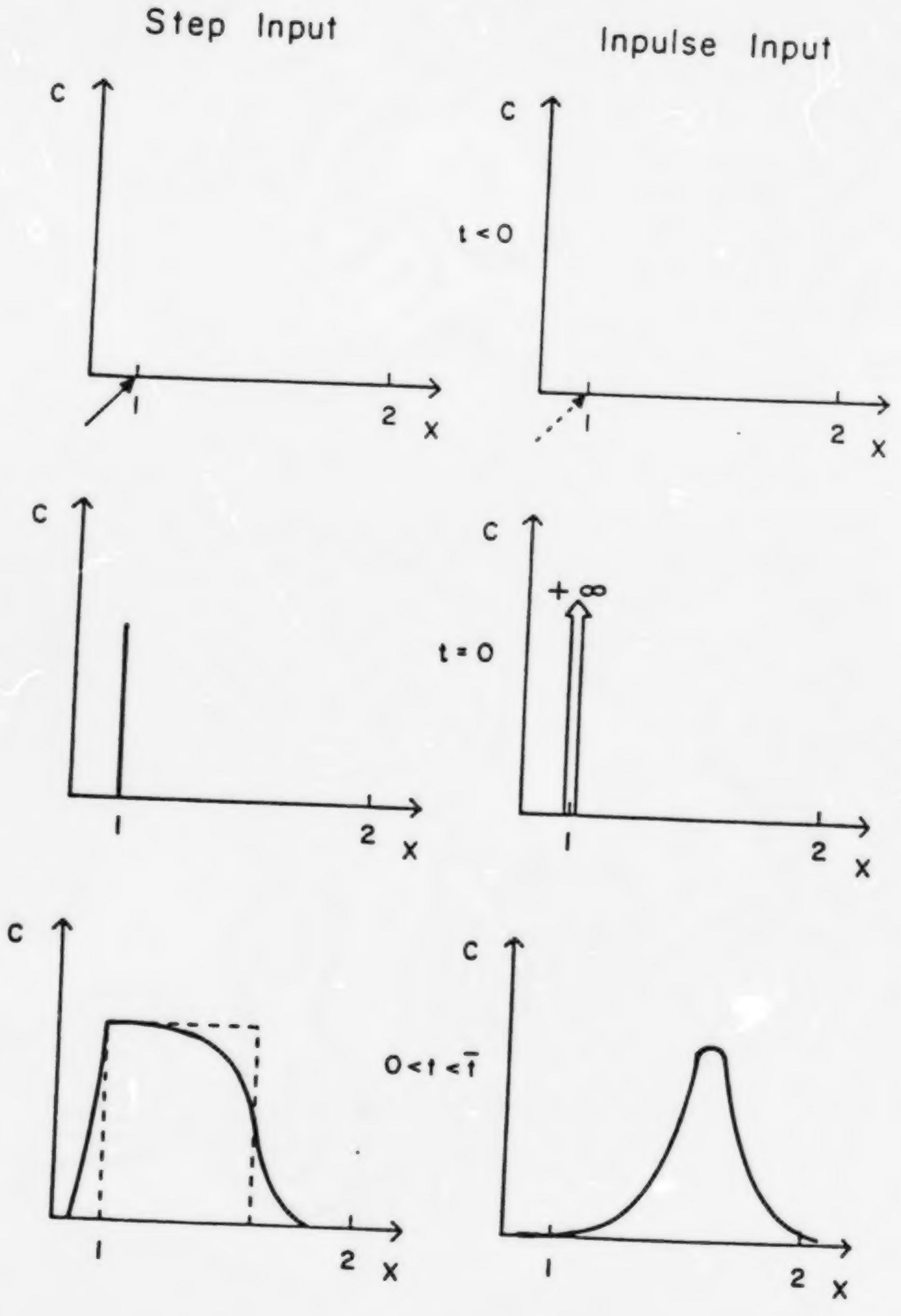
Step Input
Inpulse Input
C
t < 0
1
2
X
C
+ ∞
t = 0
1
2
X
C
0 < t < t̄
1
2
X

FIGURE D2: b. Tracer curve along the length of DPFR at various times.

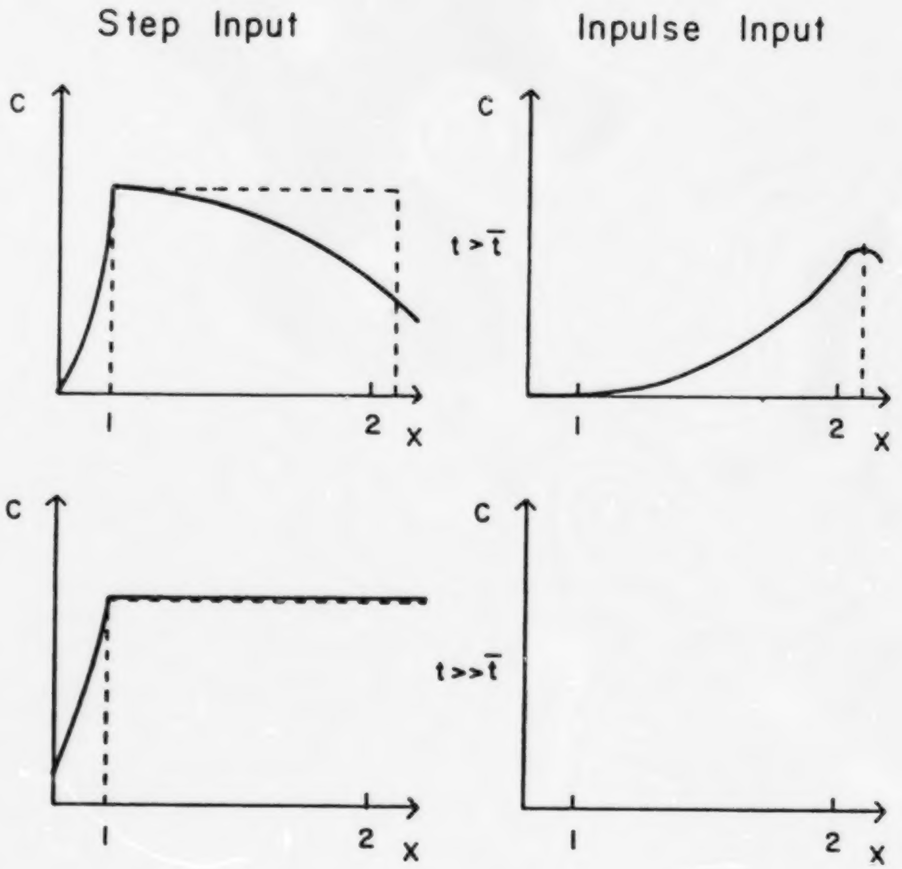

FIGURE D2: b. Tracer curve along the length of DPFR at various times (cont'd).

TABLE D1: COMPARISON OF PFR AND CSTR MODELS

Reactor Order	PFR	CSTR*
0	$C = C_o - k\bar{t}_{PFR}$	$C = C_o - k\bar{t}_{CS}$
1	$\frac{C}{C_o} = \exp(-k\bar{t}_{PFR})$	$\frac{C}{C_o} = 1 + k\bar{t}_{CS}$
2	$\frac{1}{C} = \frac{1}{C_o} + k\bar{t}_{PFR}$	$C = \frac{1}{2k\bar{t}_{CS}C_o}\left[-1 \pm \sqrt{1 + 4k\bar{t}_{CS}C_o}\right]$
n (n≠1)	$\frac{1}{C}^{n-1} = k(n-1)\bar{t}_{PFR}$	$\bar{t}_{CSTR} = \frac{1}{kC}^{n-1} (C_o/C - 1)$

* CS = CSTR

This is an amplifactor of Equation D4, but it is rephrased in terms of the total concentration of the radionuclide (C_T). Here C_T equals the sum of the soluble (C) and particulate concentrations (C_P). f is the fraction of the radionuclide which is particulate ($C_P = fC_T$), k_s is mass transfer coefficient, and the subscript s refers to sediments (e.g., ϵ_s is the porosity of sediments). All other symbols are as defined previously. On the right side of Equation D7, the first term represents inflow (see Equation D4), the second term represents losses, and the third term describes sediment-water diffusion. The second term describes three different mechanisms: outflow (see Equation D4), decay (see Equation D6) and settling (see Equation D6).

Analogous to the equations of Table D1, the solution to Equation D7 is:

$$\frac{C_T}{C_o} = \frac{1/t_w + (1-f_s)/\epsilon_s \; C_{TS}/C_o}{1/t_w + \lambda_r + f_g K_D M/\bar{z}\epsilon + (k_s/\bar{z})(1-f)/\epsilon} \qquad \text{(D8)}$$

For the DPFR model, different equations are used to estimate C/Co dependent upon the technical field from which the equations originate.

For a field in which engineered systems ("reactors") are being analyzed, the steady-state form of Equation D1 (i.e., $\delta C/\delta t = 0$) has been solved analytically for the first order reaction (n=1) by Wehner and Wilhelm (1956). The concentration C as function of position (x) and mixing levels (D/vL) is

$$\frac{C}{C_o} = \frac{2 \exp\left(\frac{vL}{D}\frac{Z}{2}\right)\left[(1+a)\cdot\exp\left(\frac{avL}{2D}(1-Z)(1-a)\exp\frac{avL}{2D}(Z-1)\right)\right]}{(1+a)^2\cdot\exp\left(\frac{avL}{2D}\right) - (1-a)^2\cdot\exp\left(\frac{avL}{2D}\right)} \qquad \text{(D9)}$$

in which

$a = 1 + 4k\bar{t}_w(D/vL)$

$Z = x/L$

D = longitudinal dispersion coefficient

v = velocity in the tank, and

L = the length of the tank.

For estuarine and river analysis, the systems geometry is abstracted such that a radionuclide is injected mid-way along the length of the DPFR and the frame of reference (x=0) is positioned at the point of discharge. The mathematical solution is given by Thomann (1972) as:

$$C = C_0' \exp(j_1 x) \text{ for } x < 0 \qquad \text{(D10)}$$

$$C = C_0' \exp(j_2 x) \text{ for } x > 0 \qquad \text{(D11)}$$

where

$$j_1 = v/2D\,(1 + (1 + 4kD/v^2)^{1/2}) \qquad \text{(D12)}$$

$$j_2 = v/2D\,(1 - (1 + 4kD/v^2)^{1/2}) \qquad \text{(D13)}$$

$$C_0' = \frac{W}{Q\sqrt{1 + \frac{4\,kD}{v^2}}} \qquad \text{(D14)}$$

W is the loading rate (M/T). Neither solution (Equations D8 or D10 to D14) considers the effect of adsorption upon migration. The solutions are equivalent and permit a similar analysis of the DPFR equation.

For groundwater analyses, the retardation concept has found some general applicability. The main idea is that non-adsorbed radionuclides move with the average linear velocity of groundwater (i.e., "non-retarded species"), whereas adsorbed radionuclides move slower (i.e., a "retarded species"). The behaviour is sketched in Figure D3. The "retarded species" moves slower than the "non-retarded species". Both concentration profiles spread as a result of dispersion.

The mean velocity of the retarded (v_c) and nonretarded (v) species are defined with respect to the $C/Co=0.5$ point on the respective concentration profiles (Freeze and Cherry, 1979). The retardation factor (RF) is defined as:

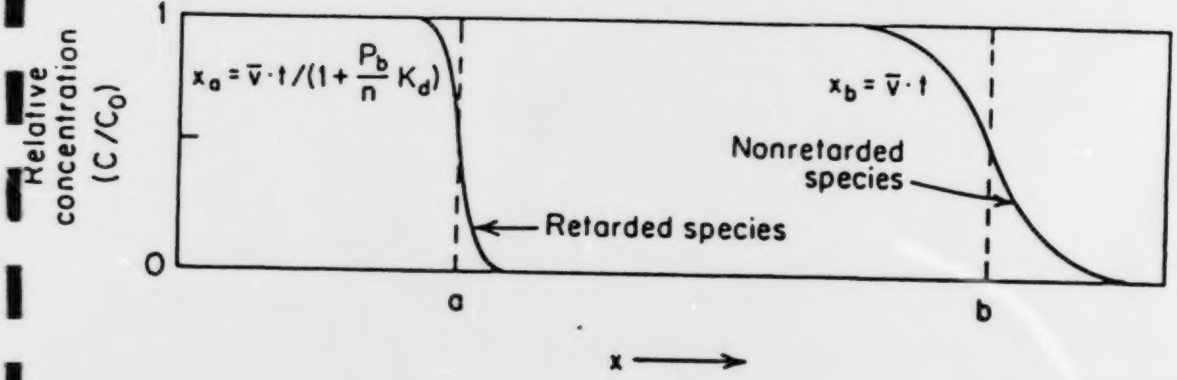

FIGURE D3: Advance of adsorbed and nonadsorbed solutes through a column of porous materials. Partitioning of adsorbed species is described by K_D. Relative velocity = $1/\ 1 + (\rho_\beta/\epsilon)K_D$. Solute inputs are at concentrations C_o at t 0. (From Freeze and Cherry, 1979; Figure 9.11)

$$\frac{v}{v_c} = 1 + \frac{\rho_\beta}{\epsilon} K_D \qquad \text{(D15)}$$

Its reciprocal, v_c/v is defined as the relative velocity.

The significance of retardation can be ascertained by considering typical characteristics of groundwater. For granular deposits ϵ is commonly in the range of 0.2 - 0.4 while the bulk mass density ρ_β ranges from 1.6 - 2.1 g/cm^3. This gives a range for the relative velocity of:

$$\frac{v_c}{v} = \frac{1}{1+4\,K_D} \text{ to } \frac{1}{1+10\,K_D} \qquad \text{(D16)}$$

The retardation concept and groundwater velocity describe transient conditions whereas the "DRL" concept describes steady state conditions. Because "DRL" is utilized only during the operating life of the facility, the question of concern is whether groundwater contamination impacts human food chains during the operating life of the facility (e.g., via wells or surface water bodies). One can make a first order determination of this by comparing the retardation velocity, the length of travel and the operating life of the facility. For example, assume that $V=100$ m/yr, $V_c=1$m/yr, length of travel to a well is 100 m and the operating life of the facility is 20 years. A nonretarded species would reach the well within 1 year, mandating its consideration for DRL purposes, whereas the 50% point of the front of the retarded species would have moved 20% of the distance suggesting that it not be considered . One should evaluate the effect of dispersion characteristics upon the profile to ensure that significant quantities of radionuclide are not dispersed ahead of the 50% point of the profile.

D.4 Transient Mixing Behaviour

The development of concepts for "Derived Release Limits" assumes steady-state behaviour. For particular mining developments, it may be desirable to conduct a field study to evalute mixing behaviour of a receiving body of water. This may evolve into a non-steady-state problem in which tracer is injected into the waterbody and mixing characteristics evaluted.

Several solutions for the transient response of a DPFR are available. For the response in the discharge end of a DPFR subjected to an impulse input of a conservative material (i.e., r=0 in Equation (1)), Murphy (1971), Thomas and McKee (1944) and Miyauchi (1953) obtained a solution to the dispersion model of the following form:

$$\frac{C}{C_o} = 2 \sum_{n=1}^{\infty} \frac{u_n(V\sin u_n + u_n \cos u_n)}{(V^2 + 2V + u_n^2)} \exp \left[\frac{V - (V^2 + u_n^2)}{2V} \right] \qquad \text{(D17)}$$

$$V = \frac{vL}{2D} = \frac{1}{2}\left[\frac{1}{D/vL}\right] \qquad \text{(D18)}$$

$$u_n = \cot^{-1}\left((u_n/V - V/u_n)/2\right) \qquad \text{(D19)}$$

$\theta = t/\bar{t}$

C_o = mass of tracer injected divided by the volume of the tank.

Here $u_1, u_2, \ldots, u_n$ are constants given by the roots of Equation D19.

Solutions for the diffusion model have been obtained using Laplace transforms. For a non-conservative, impulse input (W'), the solution is

$$C(x,t) = \frac{W'}{2A\sqrt{\pi D t}} \exp\left[- \frac{(x-vt)^2}{4Dt} - kt \right] \qquad \text{(D20)}$$

For a conservative material subjected to advection and no advection, the solutions respectively are:

$$C(x,t) = \frac{W'}{2A\sqrt{\pi D t}} \exp\left[- \frac{(x-vt)^2}{4Dt} \right] \qquad \text{(D21)}$$

and

$$C(x,t) = \frac{W'}{2A\sqrt{\pi D t}} \exp\left[- \frac{x^2}{4Dt} \right] \qquad \text{(D22)}$$

Here, A is the cross-sectional area of the tank and W' is the mass of conservative material added to the tank. By noting the similarity between Equation D22 and the form of the normal probability distribution, one may make some inferences about the mathematical characteristics of the profile of the C curve.

For a step-input of a conservative material, the mathematical solution (Ogata, 1970; Ogata and Banks, 1961) is:

$$C\,(x,t) = 0.5\ C_{BC}\ (\mathrm{erfc}\ (x-vt/2\sqrt{Dt}) + \exp\ (v/D)\ \mathrm{erfc}\ (x+vt/2\sqrt{Dt})) \qquad \text{(D23)}$$

Here C_{BC} is the concentration at the upstream boundary ($C\,(0,t) = C_{BC},\ t > 0$) and erfc = 1-erf where erf is the error function. The solution assumes that there is no material in the water body prior to application of the material and that its concentration is zero at x equal to infinity ($C\,(\infty,\ t) = 0$). For groundwater, 'l' is substituted for 'x'.

Often a combination of well mixed models can be used as a reasonable substitute for the DPFR model. Figure D4 shows a comparison of five models used for evaluating mixing from an aeration tank. The models are:

1. the dispersion model (Equation D17)
2. the well mixed model (see Table D.1)
3. the tanks in series model given by:

$$\frac{C}{C_o} = \frac{m}{(m-1)!} \quad \frac{t}{\tau_w} \quad \exp\left[-m \quad \frac{t}{\tau_w}\right] \qquad \text{(D24)}$$

   where m is the number of equal sized tanks whose total detention time is τ_w

4. the unequal tanks in series model
5. the PFR model.

Evaluation of the data led Murphy (1971) and Timpany (1966) to suggest that the dispersion model with $\frac{D}{uL}$ =0.85 gave the best fit.

The appropriate tanks-in-series models can be defined (Timpany, 1966) as

$${}^2\sigma = 2\,(D/vL) - 2\,(D/vL)^2\ (1-\exp-(vL/D)) \qquad \text{(D25)}$$

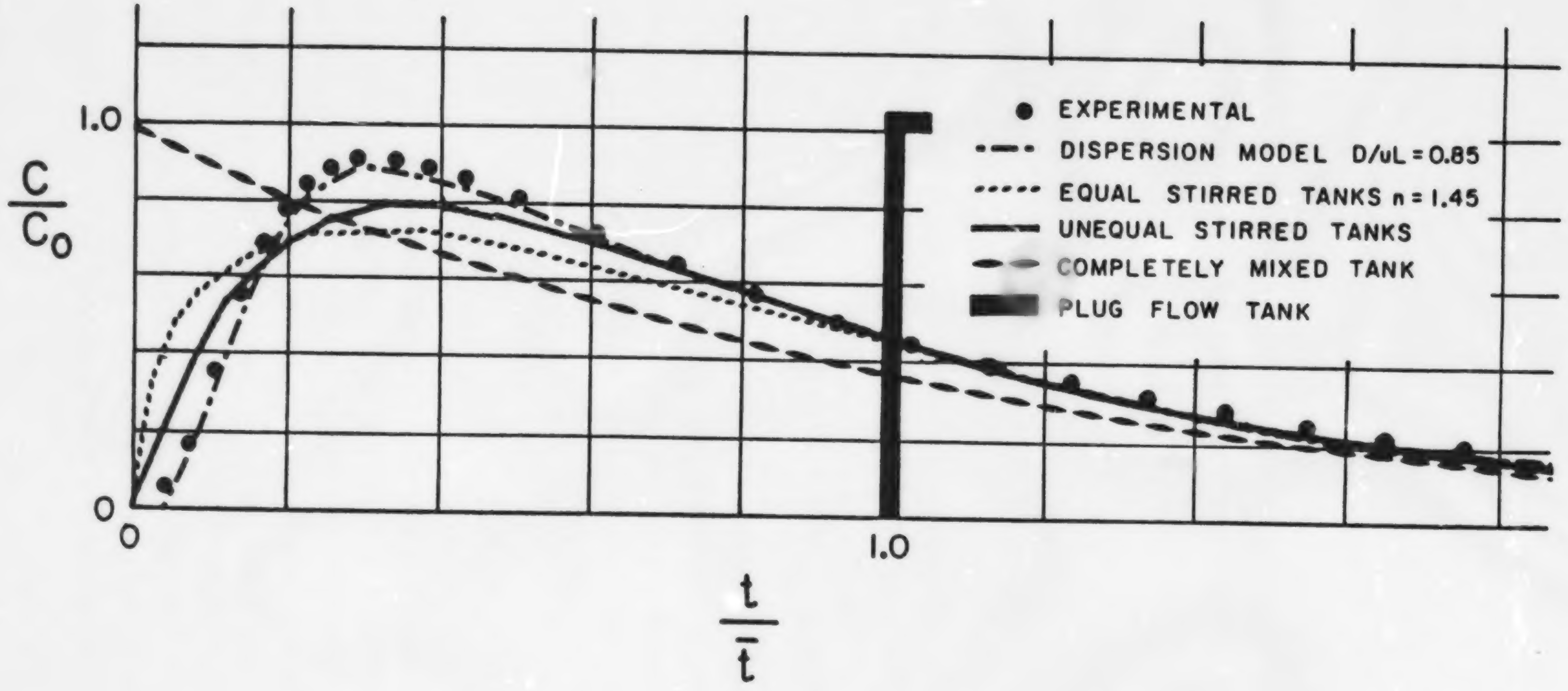

FIGURE D4: Comparison of various mixing models with observed data of Timpany (1966).

and

$$m = 1/\sigma^2 \tag{D26}$$

From these relationships one obtains $\sigma^2 = 0.701$ and $m = 1.43$. Because m is defined as an integer, an unequal tank in-series model can be obtained using the gamma distribution to estimate the noninteger factorial.

$$\Gamma(n) = (n-1)! \tag{D27}$$

Curves for the equal and non-equal sized CSTR's in series (Figure D4) show that there is reasonable agreement with the data but that the dispersion model provides the best agreement. Correlations were also obtained between the dispersion coefficient and the air flow rate in aeration tanks, by Timpany (1966).

D.5 Relationship of Dispersion Process to Other Factors

The above description of dispersion may not be sufficient for many applications because the idealizations are too strict. The dispersion process may be significant in more than one dimension (e.g., x, y). The dispersion coefficient may be a function of spatial scale (e.g., length or velocity). Numerical solutions to the equations may be most tractable. This section discusses a few of these aspects.

Multidimensional Components for Diffusion

Local variations in surface waters may often be of concern. Discharge of an effluent to a river requires a certain downstream distance before the effluent is fully mixed across the river (Figure D5a). Pronounced onshore-offshore gradients occur in all lakes (Figure D5b). For a lake of the scale of Lake Ontario, the area occupied by a plume is small, relative to the surface area of the lake but the position of a critical group on shore near the discharge is significant for DRL calculations due to spatial variations along the shore. Substantial dilution of an effluent may occur during the buoyant rise of a plume (Figure D4c). The process of mechanical dispersion is directionally dependent in an isotropic sand (Figure D5d) despite the media being isotropic with respect to hydraulic conductivity.

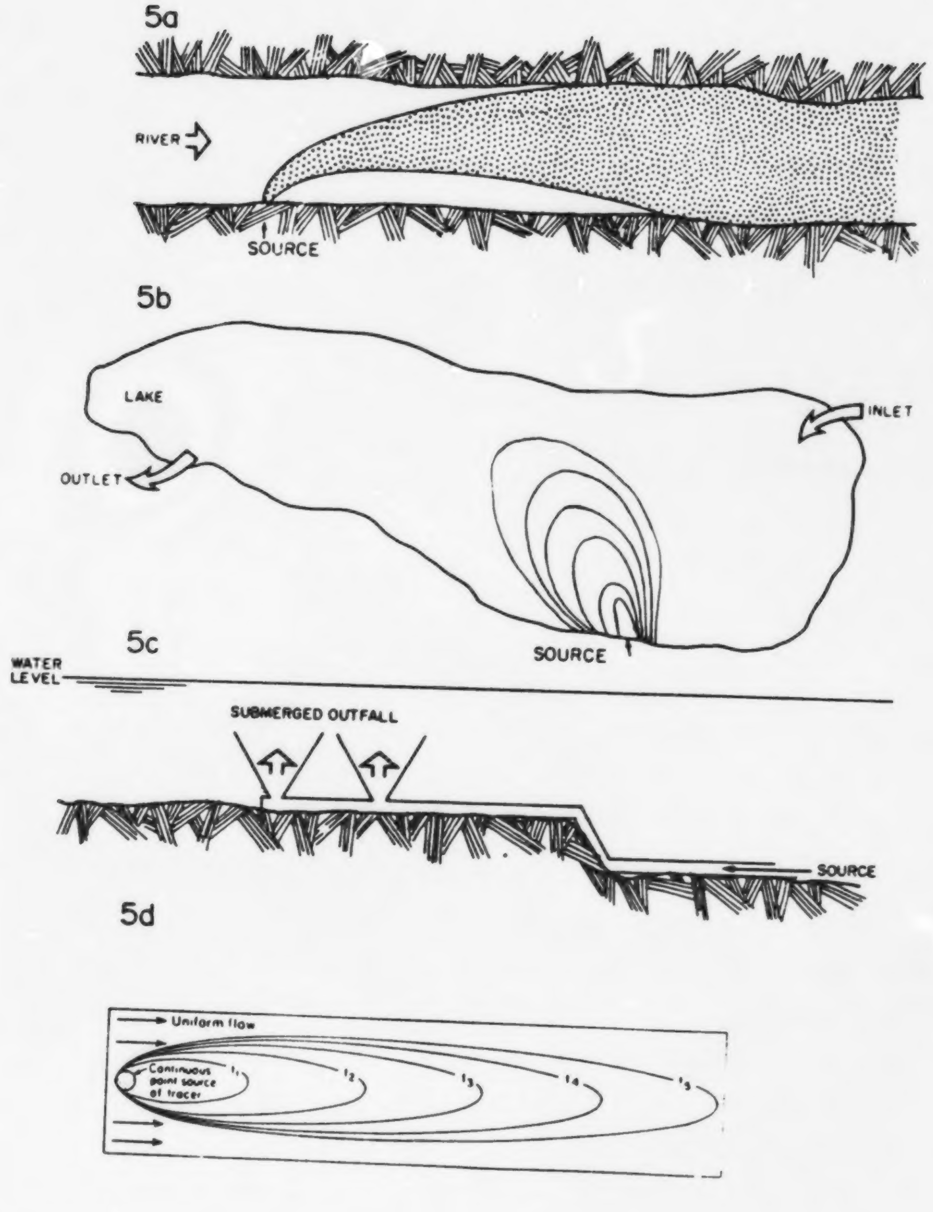
5a
RIVER
SOURCE
5b
LAKE
INLET
OUTLET
SOURCE
5c
WATER LEVEL
SUBMERGED OUTFALL
SOURCE
5d
Uniform flow
Continuous point source of tracer
t_1
t_2
t_3
t_4
t_5

FIGURE D5: Spreading of a tracer:

a. from a land-based discharge into a river

b. from shore into a lake

c. from a diffuser at the bottom of a waterbody

d. in a two-dimensional uniform flow field in an isotropic sand - continuous tracer feed with step-function initial condition (after Freeze and Cherry, 1979)

For a point source discharge to surface waters, the transport equation

$$u \frac{\delta C}{\delta x} = \Sigma_y \frac{\delta^2 C}{\delta y^2} + \varepsilon_z \frac{\delta^2 C}{\delta z^2} - kC \qquad \text{(D28)}$$

may be used (USNRC, 1977). Its solution is:

$$C = \chi \exp(-kx/u) \qquad \text{(D29)}$$

where

$$\chi = \frac{W}{2\pi u \sigma_y \sigma_z} f(\sigma_z, z, z_s, d) f(\sigma_y, y, y_s)$$

where

$$f(\sigma_z, z, z_s, d) = \sum_{m=-\infty}^{\infty} \left\{ \exp\left[-\frac{(2md + z_s - z)^2}{2\sigma_z^2} \right] + \exp\left[-\frac{(2md - z_s - z)^2}{2\sigma_z^2} \right] \right\}$$

$$f(\sigma_y, y, y_s) = \exp\left[-\frac{(y_s - y)^2}{2\sigma_y^2} \right] + \exp\left[-\frac{(y_s + y)^2}{2\sigma_y^2} \right]$$

$$\sigma_y = \sqrt{\frac{2\varepsilon_y x}{u}}, \quad \sigma_z = \sqrt{\frac{2\varepsilon_z x}{u}}$$

This transport model assumes that the input rate (W; e.g., Bq/s) and long-shore current (u) are steady. The water body is of constant depth 'd'; the shoreline is straight. Longitudinal diffusion is negligible. The origin of the frame of reference is located on shore. The discharge is located at the origin of the x-axis, a distance y_s from shoreline and z_s beneath the water surface (i.e. at (0, y_s, z_s). The lateral (ϵ_y) and vertical (ϵ_z) turbulent diffusion coefficients are assumed to be independent of scale. The removal rate, k , describes removal due to decay, settling and volatilization.

As noted by the authors (USNRC, 1977), such a model has limitations due to spatial variations in the flow field and because of large uncertainties in the values for turbulent diffusivity coefficients. Values for ϵ_y are of the order of 100-1000 cm^2/s while those for ϵ_z range from 1-30 cm^2/s (Csanady, 1970; Murthy and Csanady, 1971, Sundaram and Rehm, 1972, 1973; Gibbs and Shaw, 1977) in the near-shore region. In the thermocline, z has values of the order of 0.001-0.1 cm^2/s. The model assumes that the plume discharge is influenced only by the far field condition (i.e. that the plume is not significantly influenced by far shore line).

For specific conditions, the model can be used to develop monographs. Figure D6 shows shoreline and centreline values for the ratio χ/W as a function of distance for specific conditions. χ is the concentration of radionuclide in the water (e.g., Bq/m^3) and W is the release rate (Bq/s). The dilution factor $D_F = W/\chi Q_d$ where Q_d is the discharge rate. Neither the model nor monographs (Figure D6) include the effects of dilution during buoyant plume rise (Figure D5c). The graph (Figure D6) suggests that the downstream concentration reduction of lake plumes is rather small, consistent with observations (USNRC, 1977).

Dependence of Dispersion Coefficient (D) Upon Physics

Two aspects of physical scale are described here: the dependence of D upon scale and its dependence upon velocity. An in-depth treatment is available in standard references (e.g., Csanady, 1973).

In plumes, it is observed that ε_y depends upon length scale (cloud size) to a power which is typically four-thirds. This observation is often summarized as Richardson's four-thirds power law. It is observed in plumes both in air and surface water systems. But for systems where ε_y is reasonably constant for plume widths exceeding about 50 m, its use should be justified on a site-specific, tracer basis (Csanady, 1970; USNRC, 1977).

In shear flow, a dependency of D upon velocity is observed in surface water and groundwater systems. In streams, the dependency results from considering boundary effects upon diffusion while for groundwater the dependency results from flow around sand, grains etc.

The effects of stream boundaries upon diffusivity is represented in Figure D7, using pipe flow as the reference condition. The DPFR equation assumes that the concentration across the river, estuary, or pipe is uniform. In an actual DPFR, the concentration profile varies across the tube due to boundary effects (Condition 1; Figure D7). At higher velocities, (Condition 2 and 3; Figure D7) the cross-sectional concentration profile is more fully influenced. When the concentration gradient across this concentration profile is resolved into its normal and longitudinal components, its dependence upon velocity becomes apparent. When these phenomena are "added" to the diffusion coefficient, the resulting dispersion coefficient for a concentration profile averaged across the reactor is dependent upon the bulk fluid velocity.

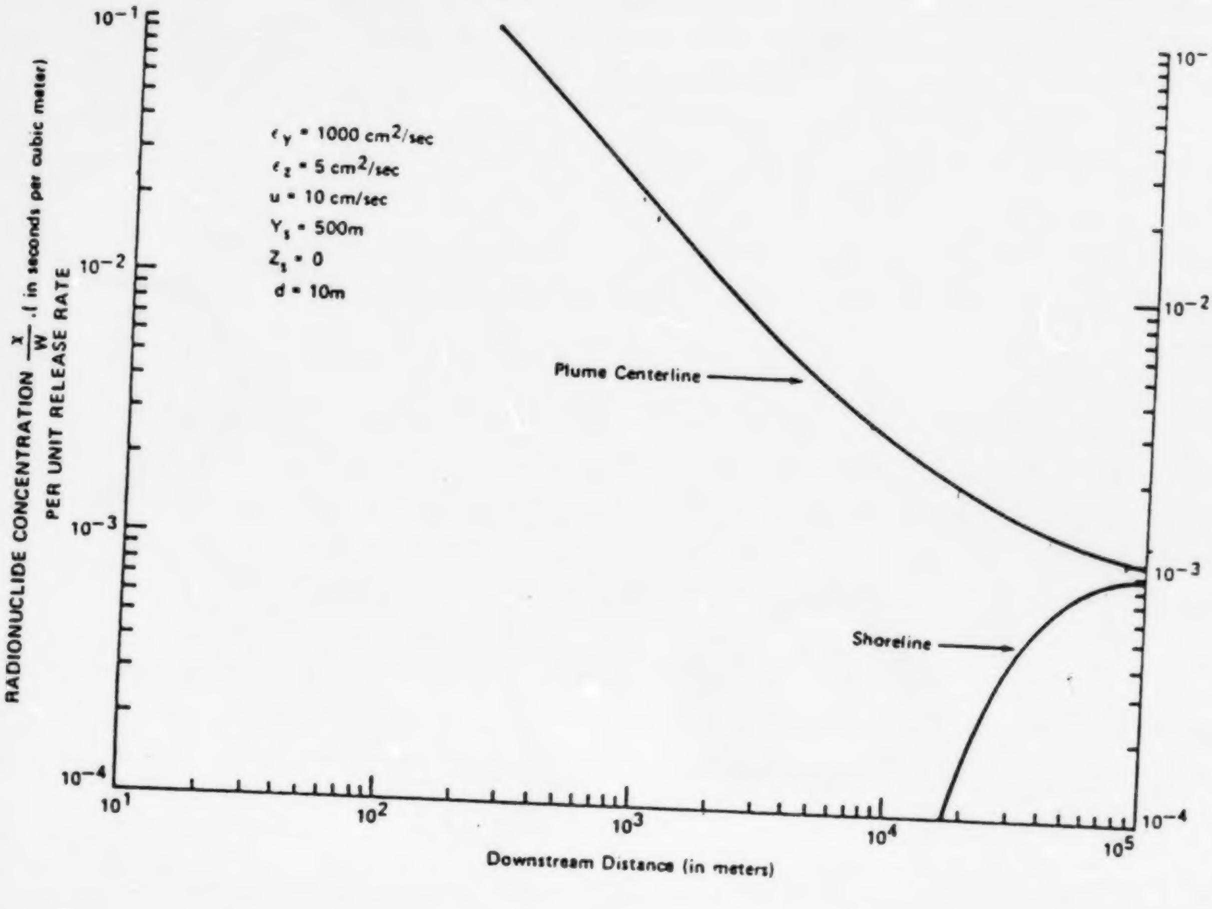

FIGURE D6: Dispersion values at surface for liquid effluents on a large lake (Figure 3; after USNRC, 1977)

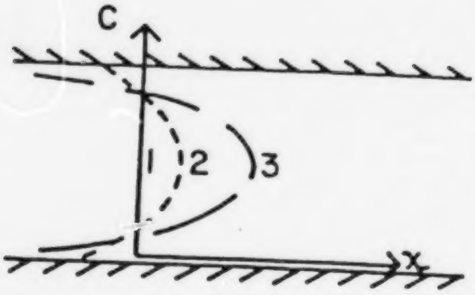

FIGURE D7: Sketch of velocity profile of DPFR.

Note:

1. idealized profile of DPFR at velocity v_1.
2. actual profile of DPFR at v_1.
3. actual profile of DPFR at velocity $v_2 (v_2 > v_1)$.

An excellent, indepth, unification of the concepts of diffusion and dispersion for surface water systems and pipes are given by Holley (1969). For groundwater systems, fluid boundary effects due to a spectrum of grain sizes and path length variation of flow around sand grains, etc. will have a significant effect upon dispersion.

With respect to terminology, care must be used with the term 'diffusivity'. Chemical engineering defines 'diffusivity' as the diffusion coefficient of a constituent in a solution (Treybal, 1973), while the groundwater field defines diffusivity as the ratio of aquifer transmissivity to aquifer storativity (Freeze and Cherry, 1979). Because the chemical engineering terminology has found wide application for other engineering and surface water systems, it is generally used in this context in this report for surface water systems (lakes, rivers, estuaries).

Models relating dispersion to hydraulic flow rate and other parameters have been proposed. For example, Fischer et al. (1979) give the relationship for the longitudinal dispersion coefficient, D_L, for an estuary of:

$$D_L = 0.011 \frac{v^2 w^2}{z\, u^*} \qquad \text{(D30)}$$

where v is the estuarine velocity, w is the width of the river; z is the depth and u* is the shear velocity at the channel bottom, $u^* = \tau_o / \rho_o$; τ_o is the shear stress and ρ_o is the fluid density. Other models are available. Fischer suggests that this relationship gives estimates within 4 times of the actual value. Hence the most accurate results will be obtained by use of the experimental data, but useful estimates can be obtained from such formulae.

For groundwater, the coefficient of hydrodynamic dispersion in the longitudinal direction D_l (see Equation D5 is expressed in terms of two components:

$$D_l = \alpha \bar{v} + D_m \qquad \text{(D31)}$$

The dynamic dispersivity (α) is a characteristic property of porous media. D_m is the molecular diffusion coefficient for the solute in the porous media. Ranges over which molecular diffusion or dispersion are most significant are shown in Figure D7. The ranges are given as a function of the Peclet Number (Pe) which is the ratio of inertial forces (velocity of flow) and diffusional forces (diffusional transport). The relationships (Figure D8) have been obtained from measurements upon saturated homogeneous granular materials.

Values of dispersitivity have generally been measured in the laboratory upon homogeneous sandy materials. Direct field measurments are rare; their values are generally inferred from modelling studies.

In the laboratory, longitudinal dispersivity is typically in the range of 0.1 to 10 mm while transverse dispersivity values are 5 to 20 times lower (Freeze and Cherry, 1979). Because the value for the transverse dispersivity coefficient is typically much lower than for the longitudinal dispersivity coefficient, molecular diffusion will control transport in a transverse direction over a much wider velocity range. In addition, transverse dispersion will be substantially smaller than longitudinal dispersion even if dispersivity is isotropic if flow velocity in the transverse direction is smaller than in the longitudinal direction. Figure D8b provides typical relationships between dispersivity and velocity for a sandstone. Note the similarity in relationships between Figure D8a and D8b.

Scaling laboratory measurements up to field scales is an area of considerable controversy. Many investigators argue that field values are larger than lab values. Some argue that dispersivity may vary as a function of the field scale being considered, while others suggest that better description of field geometry is needed Field values as large as 150 m for longitudinal dispersivity and 50 m for transverse dispersivity have been used to simulate contaminant transport in sandy aquifers (Freeze and Cherry, 1979).

D.6 Other Models

The CSA (1984) N288.1 standard has been prepared for discharges from nuclear generating stations and similar facilities. The model for aquatic systems is aggregated into a transfer function format as follows.

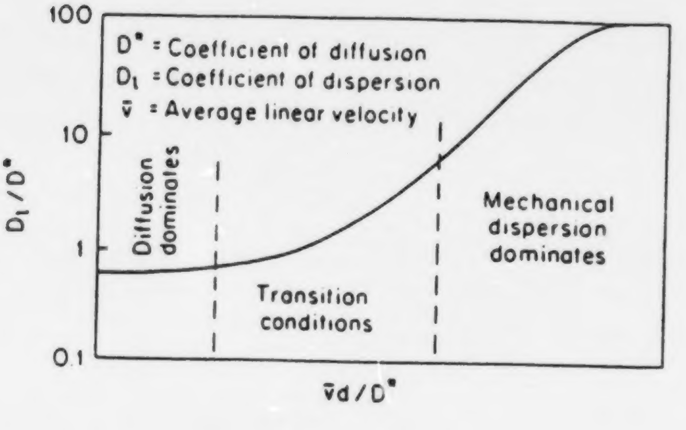

FIGURE D8a: Relation between the Peclet number and the ratio of the longitudinal dispersion coefficient and the coefficient of molecular diffusion in a sand of uniform-sized grains (after Freeze and Cherry, 1979; Figure 9.3)

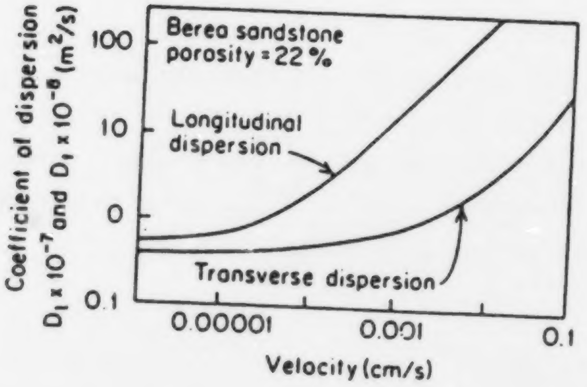

FIGURE D8b: Coefficients of longitudinal and transverse dispersion for transport in a homogeneous sandstone at various flow rates (after Freeze and Cherry, 1979; Figure 9.6)

Transfer Parameters for Releases to Surface Water - Dispersion of Liquid Effluents (P_{S2})

The transfer parameter P_{S2} relating water concentration, X_2, at a given location to the release rate, $X_S(w)$, is given by:

$$P_{S2} = \frac{X_2}{X_s(w)} = \frac{\beta}{D_F Q_v} \exp(-\lambda_d T) \qquad s \cdot L^{-1} \qquad \text{(D32)}$$

where Q_v = annual average volumetric discharge rate of the liquid effluent ($L \cdot s^{-1}$);

D_F = dilution ratio, defined as the quotient of the annual average effluent concentration at the point of discharge to the annual average concentration at the point of water use, neglecting radioactive decay;

β = effluent recirculation factor;

λ_d = $\lambda_r + \lambda_c$ (s^{-1});

λ_r = radioactive decay constant (s^{-1});

λ_c = removal constant for sedimentation (s^{-1});

T = transport time from point of discharge to point of water use (s).

The recirculation factor β allows for the build-up of radionuclide concentration in the effluent if a fraction of the effluent water is recirculated through the facility. Recirculation could occur if the water intake and discharge points are relatively close to each other.

The effluent recirculation factor is given by:

$$\beta = \frac{1}{1 - f} \qquad \text{(D33)}$$

where f = fraction of the discharge flow which recirculates back to the intake.

The term, D_F is described as a dilution ratio in the N288.1 document. It would be termed an 'inverse dilution ratio' if a more conventional definition for dilution ratio (concentration at point of use to effluent concentration) is used.

D.7 Comparison of N288.1 Transfer Parameter to Models of This Appendix

The N288.1 symbology and formula can be compared to the models for the PFR, DPFR and CSTR. The symbol $X_s(w)$ of the 'N288.1' document and the symbol 'W' used in this appendix, represent the same quantity: the rate of discharge of radionuclides to the receiving water body (units e.g., activity/day).

Re-expressing Equation D32 yields:

$$X_2 \cdot D_F = \frac{X_s(w) \exp(-\lambda_d T)}{Q_v (1-f)} \qquad \text{(D34)}$$

$$= C(\text{expec})$$

The numerator on the right-hand side represents the rate of passage of radionuclide at a time of travel, T, from the point of discharge: the rate of passage equals the rate of discharge $X_S(w)$ times the portion remaining (exp - dT) after sinks (d, describing decay, sedimentation etc.) have removed a portion of the radionuclide. The denominator of the right-hand side describes the net flow rate discharged to the environment - the actual flow rate of the discharge stream, Q_v, is reduced to $(1-f)Q_v$, because a fraction of the discharge flow (f) recirculates back to the intake. Neglecting the effects of decay, the right hand side $X_S(w)/(Q_v(1-f))$ represents the net concentration discharged to the environment. Hence the right-hand side represents the expected environmental concentration (Cexpec) after T time of travel, when dilution does not occur. The left-hand side describes the actual concentration (X_2) at T time of travel when the expected concentration, C(expec), is diluted by D_F.

A comparison of aquatic system model (i) of the N288.1 document (CDA, 1984) and (ii) those of this report (PFR, CSTR, DPFR, etc) is given in Table D2. The formulae for each model are given in Table D2, Column 2. The terms in each model are divided into the net concentration in the discharge stream (Table D2, Column 3), the dilution ratio (Table D2, Column 4) and the effects of decay (Table D2, Column 5). The net concentration terms (Column 3) are equivalent. The dilution ratios (Column 4) and the effects of decay (Column 5) are equivalent for the N288.1 derivation of P_{02} and the PFR model; the

TABLE D2: COMPARISON OF N288.1 MODEL WITH PFR, DPFR AND CSTR MODELS

Model (Col. 1)	Mathematical Expression (Col. 2)	Terms in PFR, DPFR and CSTR Model Equivalent to Those in N288.1 Model Net Concentration (Col. 3)	 Dilution Ratio (Col. 4)	 Effects of Decay (Col. 5)
N288.1	$C = (W_e \beta / Q_v)(1/D_F)e^{-\ d}$	$W_e \beta / Q_v$	$1/D_F$	$e^{-\lambda d}$
PFR	$C = C_o \exp(-k\tau_{PFR})$ $C_o = W/Q$	W/Q_v	Q_v/Q	$\exp(-k\tau_{PFR})$
DPFR	$C = C_o' \exp((v/2D)(1-(1+4kD/v^2)^{1/2})$ $C_o = W/Q\ (1+4kD/v^2)^{1/2}$	W/Q_v	$Q_v/Q(1+4kD/v^2)^{1/2}$	$\exp(v/2D(1-(1+4kD/v^2)^{1/2}))$
CSTR	$C/C_o = 1/(1 + k\tau_{CSTR})$ $C_o = W/Q$	W/Q_v	Q_v/Q	$1/(1+k\tau_{CSTR})$

Notes: W_e is the net effluent loading to the waterbody

$k = \lambda_d$

effects of decay term in the CSTR models is different from that of N288.1. For both the PFR, CSTR and the N288.1 models, dilution effects can be separated from decay effects. But in the DPFR model, the effects of dispersion, a dilution type of phenomena, are included in both the 'dilution ratio' (Column 4) and in the decay term (Column 5). The separation of dilution from decay in the N288.1 transfer parameter but not in the DPFR model can be rationalized when the method of mathematical solution for other advection-dispersion equations (e.g., Equation D29) is considered.

It is recommended here that the appropriate models (PFR, DPFR, CSTR) or their modification (e.g., as described above) be used to derive the transfer parameter P_{02}, if other than plug flow behaviour occurs in an aquatic system.

D.8 Modelling Transfer Parameter P_{28}, Surface Water to Sediments

The P_{28} values given in Appendix A (Table A11) are based upon empirical observations. These values may be derived from steady-state or non-steady-state conditions; if non-steady-state, the value applicable for DRL calculations could be substantially different. The value selected can be checked by modelling the physical process which causes transport: diffusion or sedimentation. If adsorption onto water column particulates and sedimentation is small, the steady-state form of the mass transfer model of Holloran (1982) can be used to evaluate P_{28}; this results in $g \sim 0$ in Equation D7. If adsorption and sedimentation are more important, the sedimentation term of Equation D7 can be used; this results in $k_s \sim 0$ in Equation D7.

The evaluation of these models result in the following expressions. For the case where adsorption onto water column particulates is small, P_{28} may be evaluated as:

$$P'_{28} = \frac{ks\,(1-f)/\epsilon}{ks\,(1-fs)/\epsilon_s + \lambda z_s + g_s} \qquad \text{(D35)}$$

If adsorption and sedimentation are large, then the following expression for P_{28} may be used:

$$P_{28} = \frac{g\,K_D\,Mf/\epsilon}{\lambda\, z_s + g_s} \qquad \text{(D36)}$$

Here, $f = (1 + K_D M/\epsilon)^{-1}$

$f_s = (1 + K_{DS} M_S/\epsilon_S)^{-1}$

ks = mass transfer coefficient between sediment and water ($m \cdot d^{-1}$)

f = fraction of radionuclide which is soluble in water column

f_s = value of f for sediment

K_D = adsorption coefficient in water column (cm^3/g)

K_{DS} = adsorption coefficient in sediments (cm^3/g)

M = concentration of solids in water column ($g \cdot cm^{-3}$)

M_S = concentration of solids in sediments ($g \cdot cm^{-3}$)

ϵ = porosity of water column

ϵ_S = porosity of sediments

z_S = mixed depth of sediment (m)

g = settling coefficient in water column (md^{-1})

g_s = sediment accumulation rate (md^{-1})

λ = radionuclide decay rate (d^{-1})

The expressions for sedimentation assumes that settling particles remain in place upon impaction. But shoreline sediments are often scoured and redeposited in deeper sediments; they may be replaced by relatively 'clean' sediments eroded from bluff areas. Alternatively, leaching mechanisms may cause more accumulation in deeper sediments. In such a case:

$$P_{28} = r_{sd} P'_{28} \tag{D37}$$

where r_{sd} is the ratio of radionuclide concentration in shoreline sediments to concentratin in deeper sediments. The value of r_{sd} may vary from less than 1% to 100%. In the absence of site-specific data, use of a value for r_{sd} of 1.0 is conservative.

APPENDIX E

Plume Rise and
Area Source Formulations

E.1 PLUME RISE FORMULAE

The Brigg's plume rise formulae have the general form:

$$h_r = C_1 F^{1/3} x^{2/3}/u$$

where C_1 = a dimensionless constant,
x = downwind distance (m),
u = wind speed at stack height (m/s),
F = buoyancy flux,
F = $r^2 V_s g((Tg-Ta)/Tg)$

where r = stack radius (m),
V_s = stack gas exit velocity (m/s),
Tg = stack gas exit temperature (^oK),
Ta = ambient temperature (^oK), and
g = gravitational constant (m/s^2).

The definition of C_1 is based upon the following information.

Briggs introduced different formulas for h_r depending on atmospheric stability, wind conditions and type of plumes.

i) Non-Buoyant Plumes (jets)

a) Stable Atmosphere ($\delta\theta/\delta z>0$)

Under calm conditions (u=0), the jet rises vertically without limit. h_r is given by:

$$h_r = 4\,(F_m/S)^{1/4} \qquad \text{(E1)}$$

where F_m is the moment flux parameter given as:

$$F_m = \frac{T_a}{T_g}\,\frac{V_s^2 d^2}{4} \qquad \text{(E2)}$$

where d is the stack exit diameter (m).

For windy conditions (u >0), h is given by:

$$h_r = 1.44 \ (V_s d//u)^{2/3} x^{1/3} \quad \text{(E3)}$$

The plume reaches a maximum height h_{rmax} equals to:

$$h_{rmax} = 1.5 \, (F_m/u)^{1/3} \, S^{-1/6} \quad \text{(E4)}$$

where S is a stability parameter given as:

$$S = \frac{g}{T_a} \frac{\delta \theta}{\delta z}$$

where $\frac{\delta 0}{\delta z}$ is the temperature change with height in degrees Kelvin. See Appendix C, Table C1 for values.

b) Unstable and Neutral Atmosphere $(\frac{\delta \theta}{z} \leq 0)$

For calm conditions, the jet will rise without limit. For windy conditions (u > 0), h_r is given by:

$$h_{rmax} = 1.44 \ (V_s \ d/u)^{2/3} \ x^{1/3} \quad \text{(E5)}$$

and the jet will reach a maximum height of:

$$h_{rmax} = \frac{3V_s d}{u} \quad \text{(E6)}$$

ii) Buoyant Plumes

a) Stable Atmosphere $(\frac{\delta \theta}{\delta z} \geq 0)$

Under calm conditions (u=0), the plume rise h_r is given by:

$$h_r \quad = \quad 5\,F^{1/4}S^{-3/8} \qquad \text{(E7)}$$

For u 0, Briggs indicated that h_r depends on some downwind distance x_F where the mean temperature of the plume equals the ambient temperature. x_F is given by:

$$x_F \quad = \quad \pi u S^{-1/2} \qquad \text{(E8a)}$$

For $x < x_F$, h_r is given by:

$$h_r \quad = \quad \frac{1.6F^{1/3}x^{2/3}}{u} \qquad \text{(E8b)}$$

For $x \geq x_F$:

$$h_r \quad = \quad 2.4\,(F/uS)^{1/3} \qquad \text{(E8c)}$$

b) Unstable and Neutral Atmosphere $\left(\frac{\delta\theta}{\delta z} \leq 0\right)$

Briggs divided the rise of the plume according to some downwind distance x* which, in turn, depends on F.

For $F < 55$ $\quad x^* = 14\,F^{5/8}$ (E9a)
$F \geq 55$ $\quad x^* = 34\,F^{2/5}$ (E9b)

For $x < 3.5\,x^*$ h_r is given by:

$$h_r \quad = \quad 1.6F^{1/3}x^{2/3}\,u^{-1} \qquad \text{(E9c)}$$

and for $x \geq 3.5\,x^*$

$$h_r = 1.6F^{1/3}(3.5x^*)^{2/3}\,u^{-1} \qquad \text{(E9d)}$$

Most computer models utilize only the final plume rise from the Briggs equations. For unstable or neutral conditions (Stability Classes A, B, C and D), the downwind distance of final plume rise is defined by:

$$x_F = 3.5\,x^*$$

and x^* is defined by Equations E9a and b, and h_{rmax} is defined by Equation E9d.

For stable conditions (Stability Classes E and F), the downwind final rise is defined by Equation E8a, with $\delta\theta/\delta z$ equal to 0.015 ^{o}K/m for Stability E and 0.035 ^{o}K/m for Stability F. The final plume rise is then given by Equations E7 for near calm conditions and E8C for windy conditions.

For the final plume rise under these conditions, the smaller of the values from Equations E7 and E8c is used.

The buoyant plume form of the equations should be used when the stack exit temperatures are above ambient. Having applied the Briggs formulae, it is possible to determine the effective plume rise and increment the source height to interpolate the data from Figures A1-A6. Default values for various plume rise situations may be interpolated from Figures E1 and E7.

E.2 AREA SOURCE CONSIDERATIONS

E.2.1 Area Source General Treatment

This treatment follows that utilized in the MILDOS model (NUREG/CR 2011, 1981).

The sources at the site should be reviewed to define areas of similar emissions, i.e., tailings, storage areas, haulage routes, waste rock, storage, etc. These areas should be subdivided into subsources with dimensions that provide a maximum size of 300 m by 300 m squares. For population exposure calculations within 80 km, the concentrations are evaluated at the midpoint of each spatial interval, and applies to all points within that sector. However for individual receptor locations the concentration is weighted by the distance from the midline of the sector. This is calculated by adjusting the source strength to be:

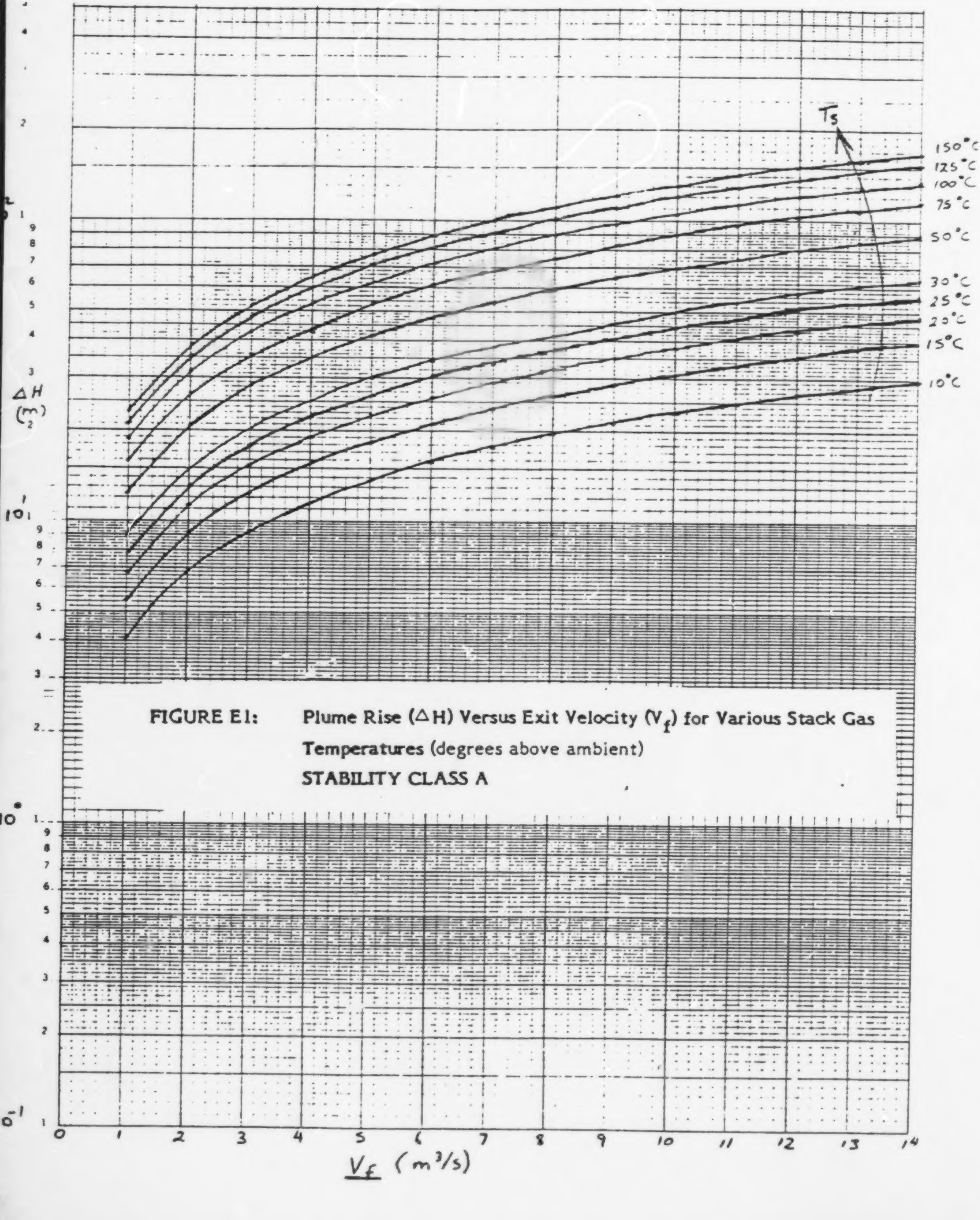

FIGURE E1: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASS A

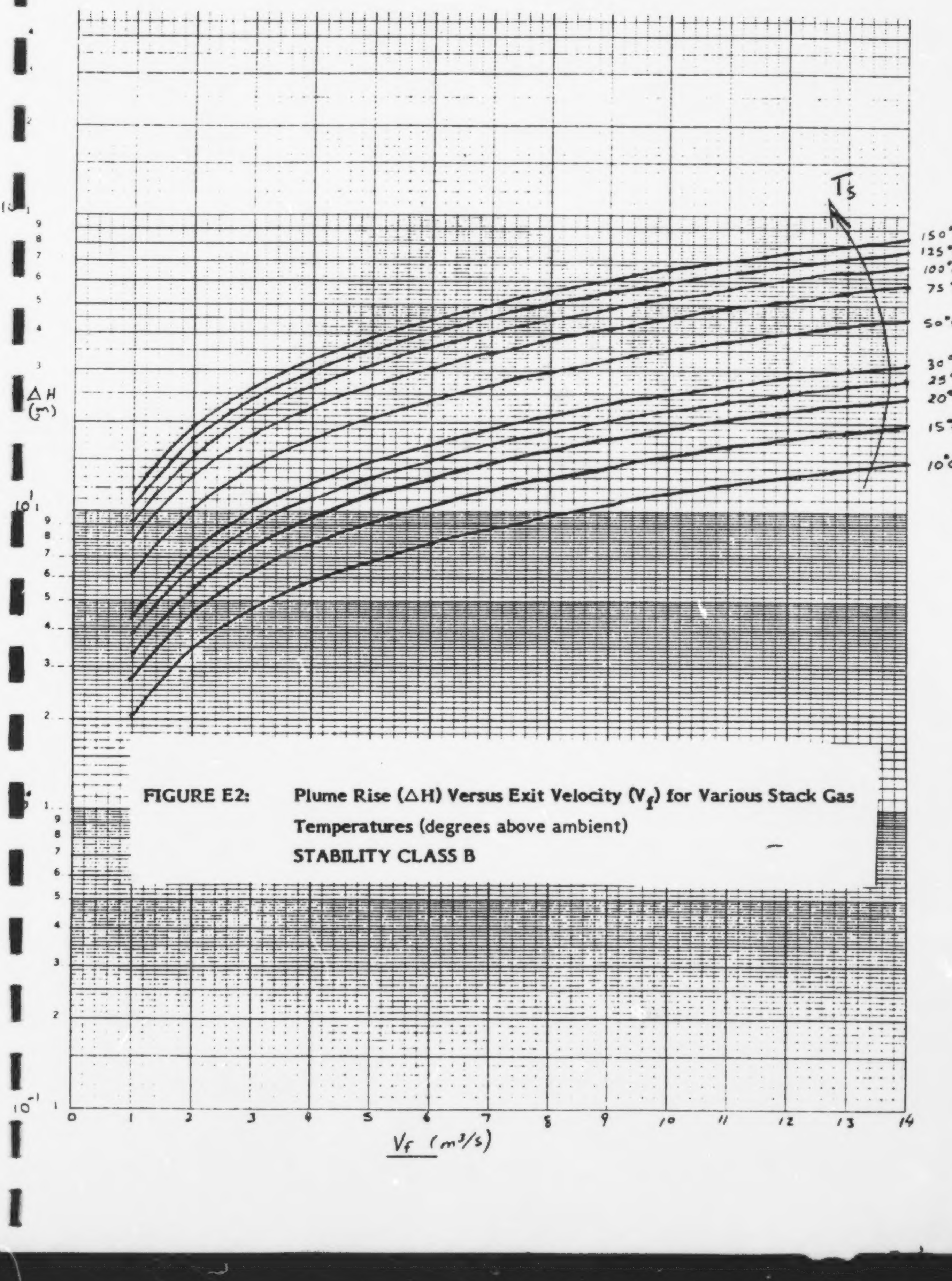

FIGURE E2: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASS B

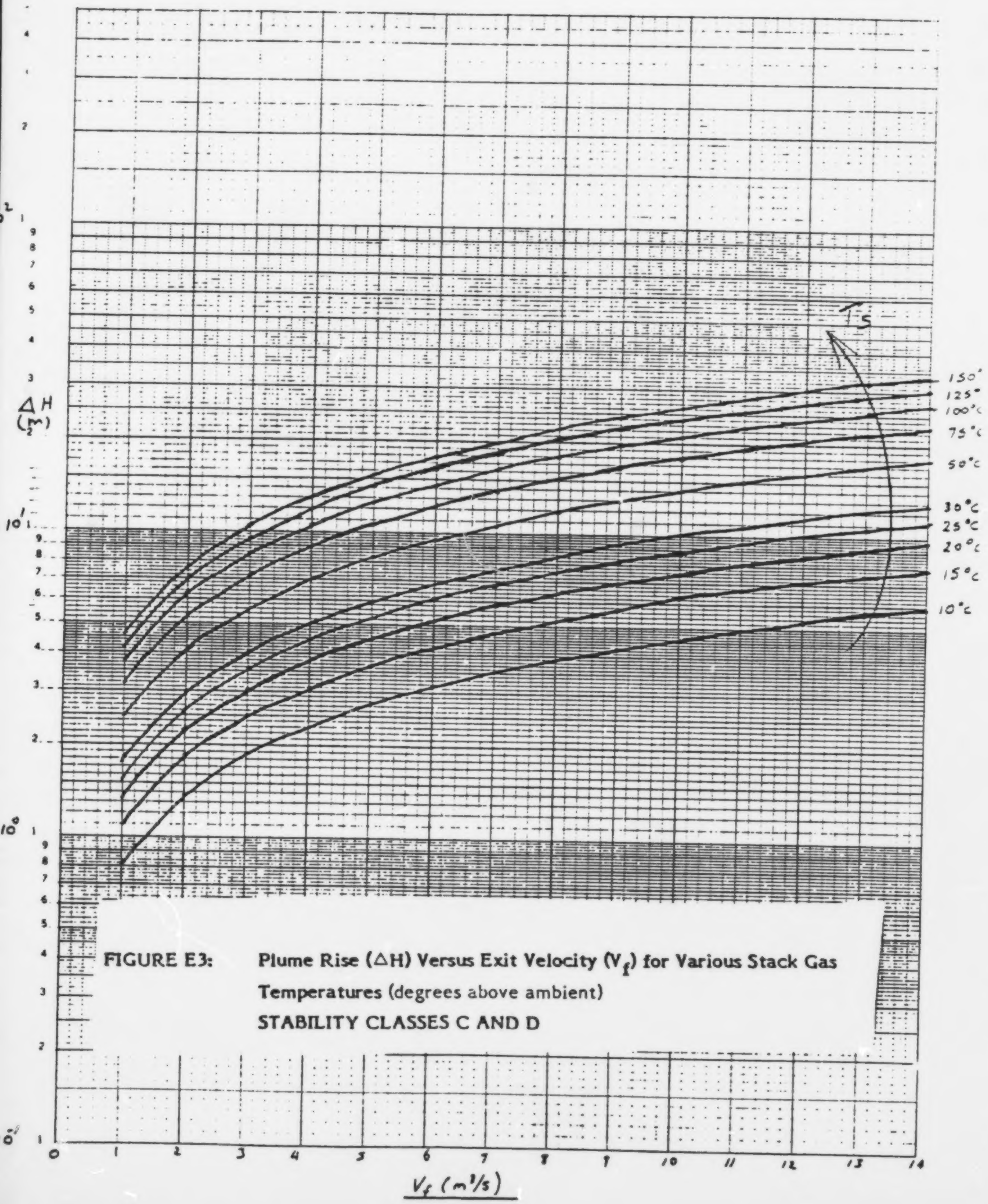

FIGURE E3: Plume Rise (ΔH) Versus Exit Velocity (V_s) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASSES C AND D

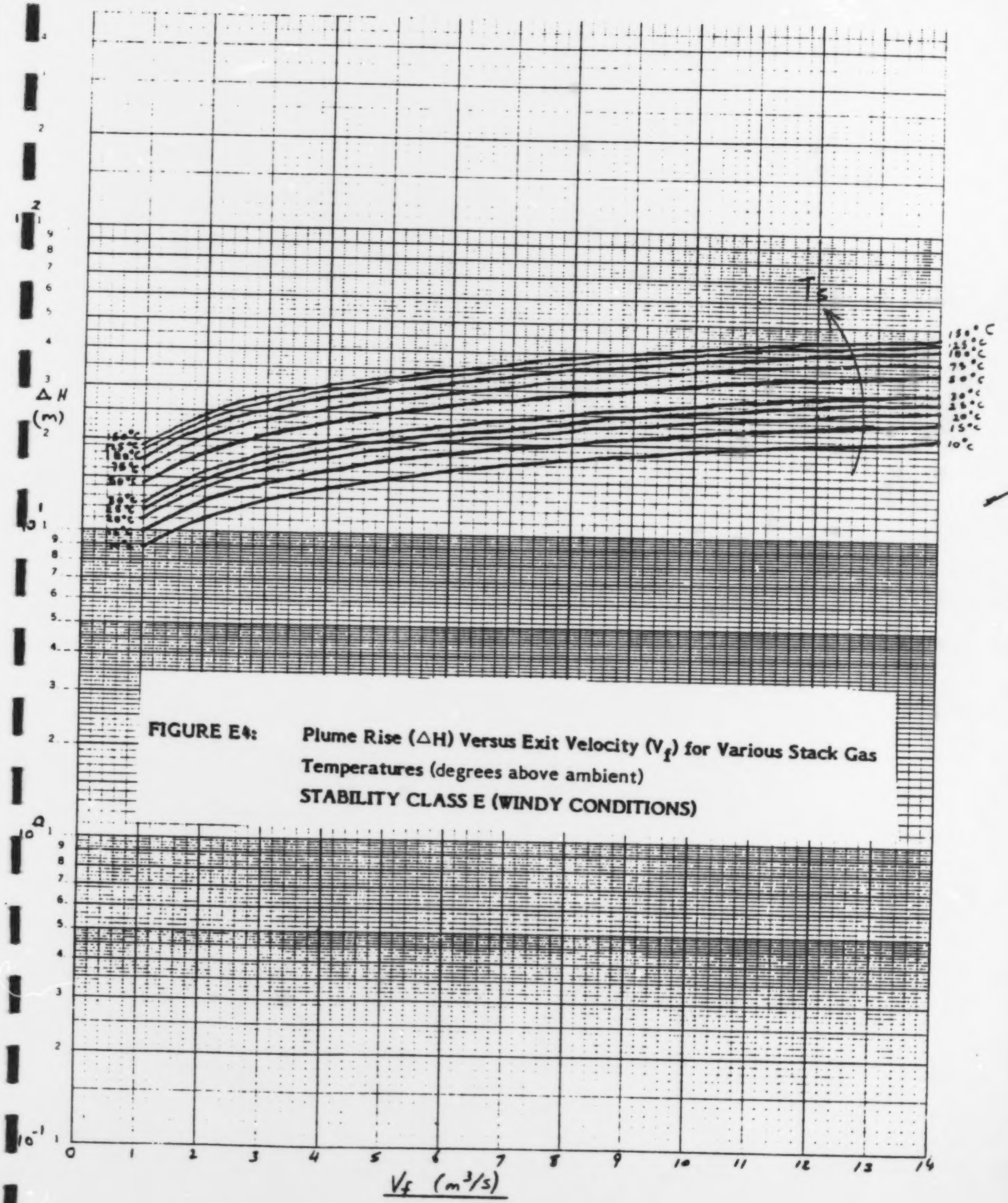

FIGURE E4: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)

STABILITY CLASS E (WINDY CONDITIONS)

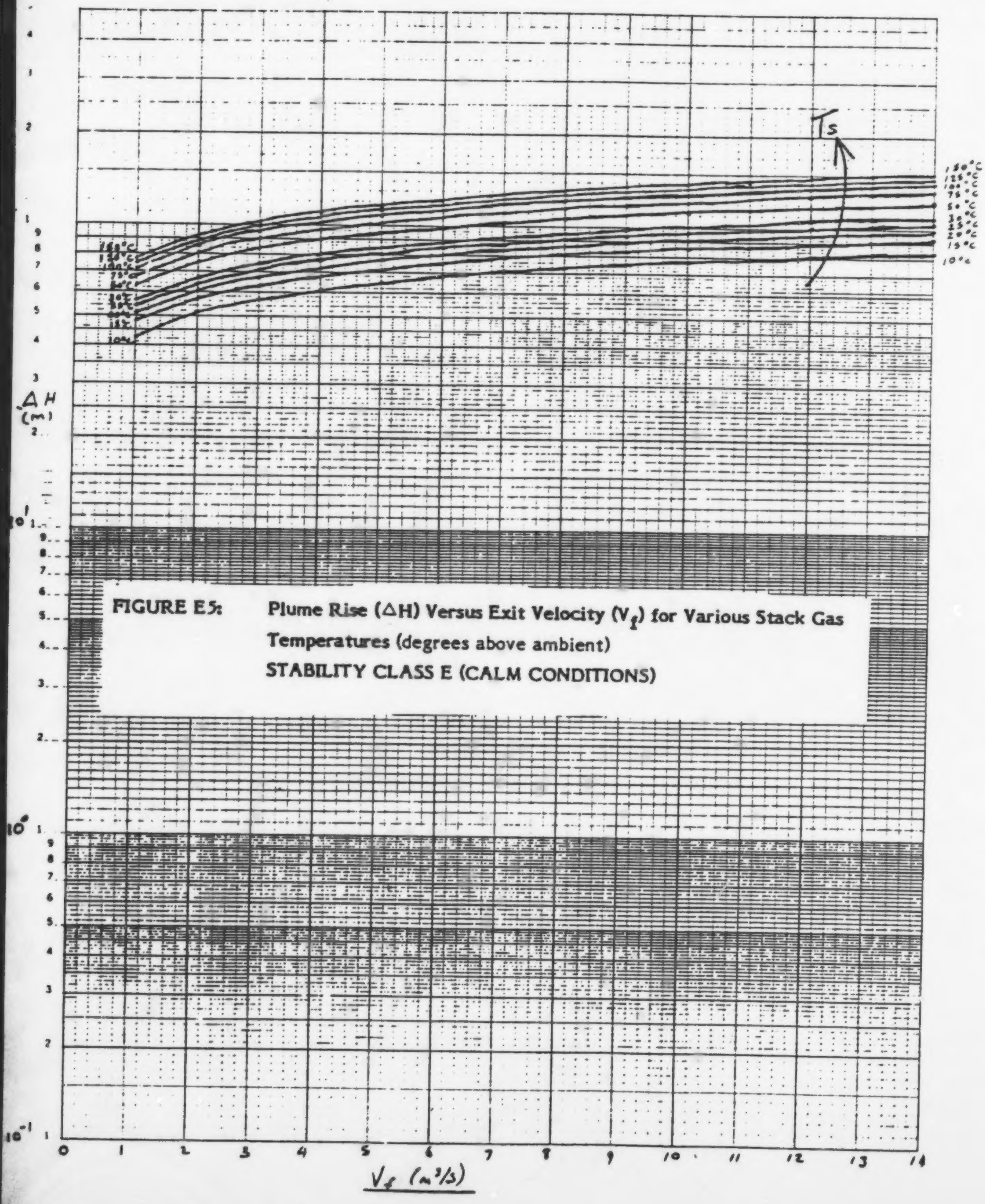

FIGURE E5: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASS E (CALM CONDITIONS)

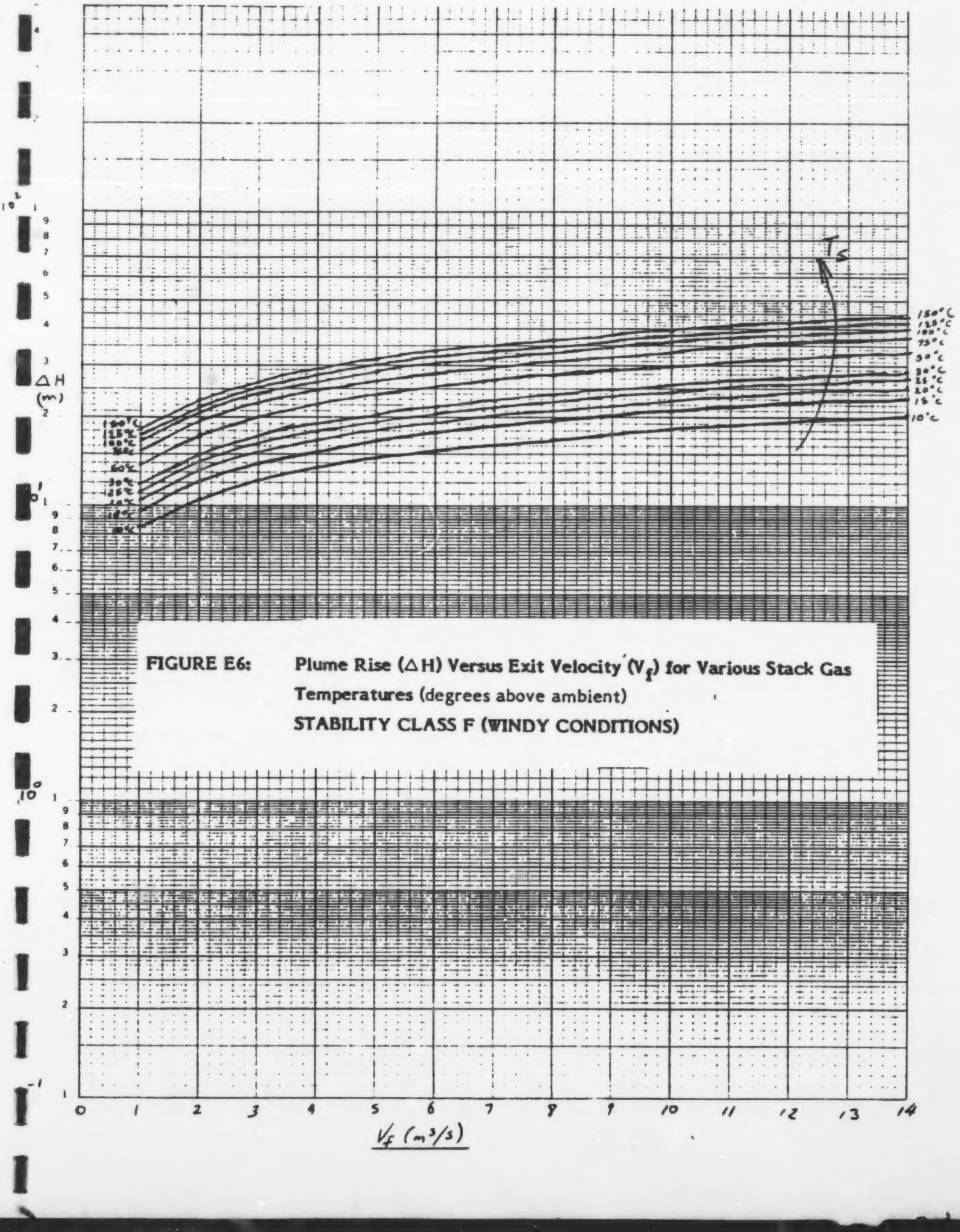

FIGURE E6: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASS F (WINDY CONDITIONS)

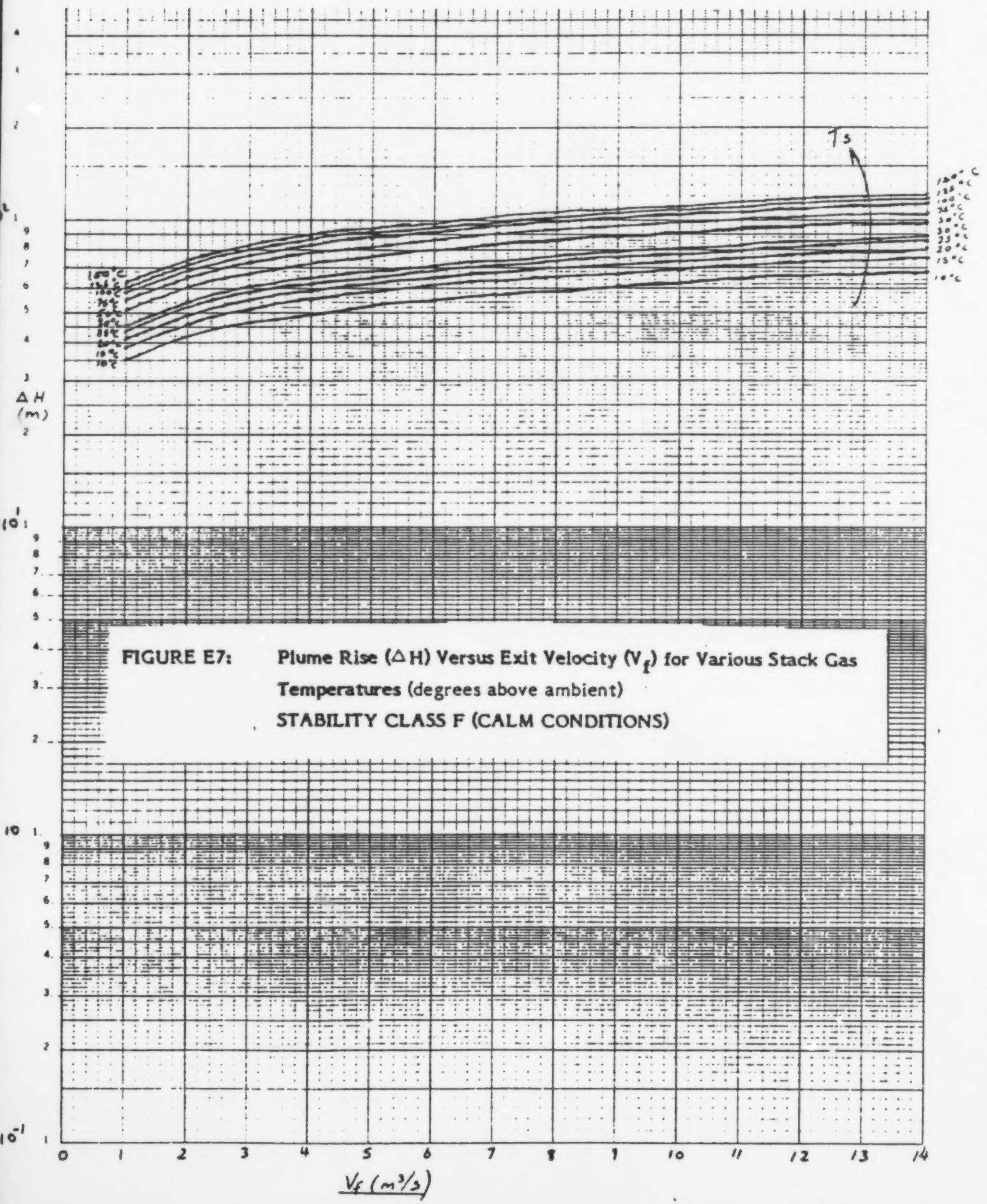

FIGURE E7: Plume Rise (ΔH) Versus Exit Velocity (V_f) for Various Stack Gas Temperatures (degrees above ambient)
STABILITY CLASS F (CALM CONDITIONS)

where: $$Q(i,j,y) = \frac{\left(\frac{\pi}{8}x-y\right)}{\frac{\pi}{8}x} Q(i,j,o) \qquad \text{E10}$$

Q = emission rate Bq/s

i = pollutant -Rn

j = source (tailings)

y = cross wind dist.

x = down wind dist.

o = centerline conc.

Thus any receptor within 22-1/2° (one sector width) in either direction from the centerline of the wind direction sector receives some concentration. All other receptors receive zero. Area sources are given an additional lateral dispersion by using a "virtual point source" method. The distance to the "virtual point", taken upwind from the area source is calculated by:

where: $$X_{vp} = \frac{8}{\pi} S \qquad \text{E11}$$

This equation is defined in Equation 5.4.

The calculation of downwind distance to the receptor is defined by:

$$x' = x + x_{vp} \qquad \text{E12}$$

where x' is the area source distance downwind,

x is the distance from area centre to receptor,

x_{vp} is the virtual point source displacement distance.

E.2.2 Narrow Plume Correction

Area sources may not give rise to the same concentration at all receptors. Not only do the relationships of Equation 5.2 govern the values, but some receptors may not get the full effect of the source. This situation requires some adjustment of the source strength so close-in receptors do not provide unrealistically high transfer parameters.

NUREG/CR2011, 1981) applies the narrow plume hypothesis compensation developed by TRW Inc. for the AQDM model.

This procedure is based upon the following derivation originally proposed by Gifford (1959). It is assumed that the variable area-source contribution obeys the steady-state diffusion equation in two independent variables:

$$u(z)\,(\delta\chi_A/\delta x) = (\delta/\delta z)\,K(z)\,(\delta\chi_A/\delta z) \qquad \text{(E13)}$$

where u (z) is the mean wind velocity ($m \cdot s^{-1}$) in the x-direction, z is height above the ground surface (m) and $_A$ is the area source contribution to the downwind concentration (Bq/m^3). In this equation, K defines the eddy-diffusivity. This approach neglects the y-component of diffusion. This is equivalent to the observation that point source plumes in the atmosphere tend to be long and narrow. Thus, the concentration at a point can be influenced only by sources in a narrow, plume-shaped upwind sector (Gifford and Hanna, 1971). This assumption allows one to approximately replace the two-dimensional integral that expresses the total area-source contribution to concentration at a receptor by a one-dimensional integral that can be analytically defined. The integral involves only two things:

- a knowledge of the distribution of the area-source emission rates along the line in the direction of the upwind azimuth from the receptor location, and
- the meteorologically dependent function that specifies the crosswind-integrated concentration in the Gaussian plume from a point source.

This approach has been found to agree quite well with other more complex approaches to area source modelling. The application of the approach involves the use of a utilization factor (Gifford and Hanna, 1971).

This factor corrects for the condition of close-in receptors only seeing part of the emissions. The utilization factor is the ratio of that proportion of source area lying within 22.5° sector upwind of receptor to the total area.

APPENDIX F

Radon and Radon Daughters
Special Quantities and Units

F.1 RADON-222 AND DAUGHTERS

Radon-222 is a gas that can be transported in soil gas, in water, and by air. It is slightly soluble in body fluids, and more soluble in fats. People exposed to it in air will receive a relatively small dose, fairly uniformly distributed throughout body tissue from this dissolved radon. Of more importance is that radon-222 is the precursor of the short-lived "radon daughters". These isotopes (Ra-A (Po-218), Ra-B (Pb-214), Ra-C (Bi-214) and Ra-C' (Pb-214)), when formed in air become attached to particles or remain as free atoms (unattached). When inhaled, they deposit preferentially on the surfaces of the bronchial tree, resulting in relatively high doses to cells of the bronchial epithelium. The dose to the pulmonary region of the lung is about a factor of 8 below that to the bronchial epithelium, and that to other tissues of the body is insignificant.

The effect from exposure to radon daughters in air is measured in terms of the potential alpha energy from their decay to Pb-210. The traditional unit for this measurement is the working level (WL), with the working level month (WLM) the unit of accumulated exposure. The WL is defined as any concentration of the short-lived daughters in one litre of air that will result in 1.3×10^5 MeV of alpha energy in their decay to Pb-210. The WLM is 170 times this quantity (170 being the assumed hours worked per month). The SI equivalents for these units are:

$$1\ \mathrm{J \cdot m^{-3}} = 4.80 \times 10^4\ \mathrm{WL}$$
$$1\ \mathrm{J \cdot h \cdot m^{-3}} = 285\ \mathrm{WLM}$$

One WL corresponds to the potential alpha energy concentration of short-lived radon daughters in air which are in radioactive equilibrium with a Rn-222 concentration of 3.7 kBq $\mathrm{m^{-3}}$. Hence:

$$1\ \mathrm{J \cdot m^{-3}} = 1.78 \times 10^8\ \mathrm{Bq \cdot m^{-3}}\ \text{Rn-222 in radioactive equilibrium with its short-lived daughters.}$$

The ICRP (1981) has recently reviewed the available dosimetry and epidemiology data for radon daughters and has recommended 0.017 $\mathrm{J \cdot h \cdot m^{-3}}$ as the annual limit for occupational exposure of atomic radiation workers (ARW). This value implies that an exposure of one $\mathrm{J \cdot h \cdot m^{-3}}$ results in an effective dose equivalent of 2.94 Sv, in good agreement with values calculated by an NEA Working Group (NEA, 1983). The ICRP has not recommended limits for non-occupational exposures, but the NEA Working Group, taking lung size and breathing rates, and ranges of expected unattached fractions and particle sizes into account, has recommended a dose conversion factor of 1.5 Sv per $J \cdot h \cdot m^{-3}$ for all ages (NEA, 1983).

F.2 RADON-220 AND DAUGHTERS

Radon-220 (thoron) has a much shorter half-life than radon-222 (radon) and, therefore, is usually much less "transportable" in the environment. It also has short-lived daughters (Th-A (Po-216), Th-B (Pb-212) and Th-C (Bi-212)) which also result in high doses to the bronchial epithelium if inhaled. In addition, the longer half-life of the thoron daughters as compared to radon daughters results in a significant dose to tissues other than those of the lung.

The NEA Working Group (NEA, 1983) also reviewed the dosimetry of thoron daughters and has recommended a conversion factor of 1.0 Sv per $J \cdot h \cdot m^{-3}$ for all occupational exposures to thoron daughters, and of 0.5 Sv per $J \cdot h \cdot m^{-3}$ for all non-occupational exposure (NEA, 1983).

F.3 SUMMARY

Dose conversion factors for radon and thoron daughters are given in Table F1. In the table, "average member of the public" represents the individual who engages in 16 hours of light activities and 8 hours of resting per day. The dose factor may be applied to all age groups.

TABLE F1: DOSE CONVERSION FACTORS FOR RADON AND THORON DAUGHTERS

	Dose Conversion Factors	
	"Light Activity" Workers (non-ARW)*	Average Member of the Public, All Ages
Radon Daughters		
$Sv/J \cdot h \cdot m^{-3}$	2.94	1.5
$Sv \cdot a^{-1}/(J \cdot m^{-3})$	-	1.3×10^4
Sv/WLM	1.03×10^{-2}	5.5×10^{-3}
Thoron Daughters		
$Sv/J \cdot h \cdot m^{-3}$	1.0	0.5
$Sv \cdot a^{-1}/(J \cdot m^{-3})$	-	4.4×10^3
Sv/WLM	3.5×10^{-3}	1.8×10^{-3}

*Assume worker works eight hours per day.

APPENDIX G

Sample Calculations

APPENDIX G: SAMPLE CALCULATIONS

This appendix illustrates DRL calculations for an atmospheric release and an aqueous release, based on the methodologies described in the text of this report and on the default values given in Appendices A to C.

It is perhaps most logical to conceptualize the calculation of a DRL from source release, through the appropriate pathways, to dose. This is the manner in which most users of this report will probably make a DRL calculation if a simple desk-top calculator is used. If the user wishes to make several DRL determinations to evaluate the effects of varying critical group habits or transfer parameter values, the pathways models may be computerized, and programmed to run "backwards" from dose to source release, as illustrated in Figure G1. In this fashion, the dose may be set at the annual dose limit or some fraction thereof, and the effects of adjusting various portions of the pathways model values on the DRL may then be observed. In so doing, the user may manipulate the model by selecting source term reduction options to optimize the dose reduction in terms of economic considerations.

In the following sample calculations, releases have been selected which result in a dose equal to the annual dose limit, i.e., the release rates are equal to the derived release limits.

G1 SAMPLE CALCULATION OF DOSE FROM ATMOSPHERIC RELEASES OF RADON-222

In this example, we calculate an atmospheric emission DRL 9.5×10^8 $Bq \cdot s^{-1}$ of Rn-222 from milling operation. The radon is assumed to be released from an average emission height of 20 m. It is also assumed that the frequency of each atmospheric stability class is as shown in Table C1.

A critical group inhabits an area 5 km from the milling operation. Average Rn-222 dilution factors (P_{S1} values) for each stability category, ignoring radioactive decay, may be read from the graphs provided in Figures A1-A6:

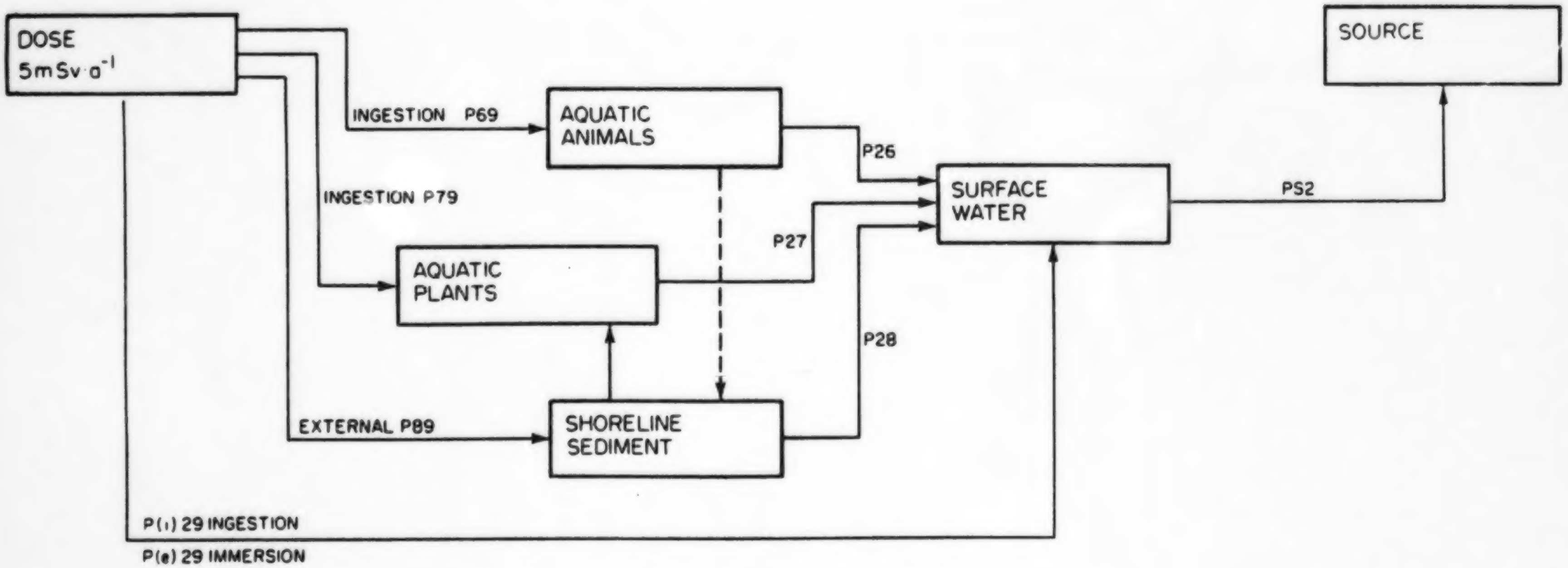
DOSE
5mSv·a⁻¹
INGESTION P69
INGESTION P79
EXTERNAL P89
P(i) 29 INGESTION
P(e) 29 IMMERSION
AQUATIC ANIMALS
AQUATIC PLANTS
SHORELINE SEDIMENT
P26
P27
P28
SURFACE WATER
PS2
SOURCE

FIGURE G1: CALCULATION DERIVED RELEASE LIMIT FROM DOSE TO SOURCE

Stability Class	P_{S1} (s.m^{-3})
A	3.5×10^{-8}
B	3.4×10^{-8}
C	2.1×10^{-8}
D	3.0×10^{-8}
E	9.2×10^{-8}
F	2.4×10^{-7}

The overall average P_{S1} value, weighted according to the stability class frequency shown in Table C1, is 7.1×10^{-8} s.m^{-3}.

With a release of 9.5×10^{8} Bq.s^{-1} of Rn-222, the atmospheric concentration at the critical group receptor may be calculated as:

$$\begin{aligned} X_1 &= X_S \cdot P_{S1} \\ &= 9.5\times10^{8}\ \mathrm{Bq\cdot s^{-1}} \times 7.1\times10^{-8}\ \mathrm{s\cdot m^{-3}} \\ &= 6.74\times10^{1}\ \mathrm{Bq/m^{3}} \end{aligned}$$

As a gas, Rn-222 does not pass through soil, vegetation and animal pathways, but is usually of primary concern due to the high dose conversion factors for inhalation associated with Rn-222 and its short-lived daughters. Thus, only $P(i)_{19}$ and $P(e)_{19}$ are considered here.

G1.1 $P(i)_{19}$ Atmosphere to Dose (Inhalation)

$$P(i)_{19} = 5.6\times10^{-9}\, \phi (OF)_i \cdot (DCF)_{Rn}\ \text{daughters}\ \mathrm{Sv\cdot a^{-1}\cdot Bq^{-1}\cdot m^{3}}$$

Assuming that $\phi = 1$, $(OF)_i = 1$, and $(DCF)_{Rn}$ daughters = 1.3×10^{4} Sv$\cdot$a$^{-1}\cdot$J^{-1}m^{3}

$$P(i)_{19} = 7.2\times10^{-5}\ \mathrm{Sv\cdot a^{-1}\cdot Bq^{-1}\cdot m^{3}}$$

Thus, an average air concentration of 6.74×10^{1} Bq$\cdot$m^{3} gives rise to an inhalation dose as follows:

$$
\begin{aligned}
X(i)_9 &= X_1\ P(i)_{19} \\
&= 6.74\text{x}10^{1}\ \text{Bq}\cdot\text{m}^{-3}\ \text{x}\ 7.2\text{x}10^{-5}\ \text{Sv}\cdot\text{a}^{-1}\ \text{Bq}^{-1}\cdot\text{m}^{3} \\
&= 4.85\text{x}10^{-3}\ \text{Sv}\cdot\text{a}^{-1}
\end{aligned}
$$

G1.2 $P(e)_{19}$ Atmosphere to Dose (Immersion)

$$P(e)_{19} = (f_u + (1 - f_u)\ S)\ (DCF)_a$$

Assuming that $f_u = 0.2$, $S = 0.9$ and $(DCF)_a = 2.53\text{x}10^{-6}\ \text{Sv}\cdot\text{a}^{-1}\cdot\text{Bq}^{-1}\cdot\text{m}^{3}$, then

$$
\begin{aligned}
P(e)_{19} &= (0.2 + (1\text{-}0.2)\ 0.9)\ 2.53\text{x}10^{-6}\ \text{Sv}\cdot\text{a}^{-1}\cdot\text{Bq}^{-1}\cdot\text{m}^{3} \\
&= 2.4\text{x}10^{-6}\ \text{Sv}\cdot\text{a}^{-1}\cdot\ \text{Bq}\cdot\text{m}^{3}
\end{aligned}
$$

Thus, an average air concentration of $6.63\text{x}10^{1}\ \text{Bq}\cdot\text{m}^{-3}$ gives rise to an external dose as follows:

$$
\begin{aligned}
X(e)_9 &= X_1\ P(e)_{19} \\
&= 6.74\text{x}10^{1}\text{Bq}\cdot\text{m}^{-3}\ \text{x}\ 2.4\text{x}10^{-6}\ \text{Sv}\cdot\text{a}^{-1}\cdot\text{Bq}^{-1}\text{m}^{3} \\
&= 1.6\text{x}10^{-4}\ \text{Sv}\cdot\text{a}^{-1}
\end{aligned}
$$

G1.3 Total Dose from Rn-222 Through Atmospheric Pathways

The total dose from Rn-222 from atmospheric pathways is $X(i)_9 + X(e)_9 = 5.0\text{x}10^{-3}\ \text{Sv}\cdot\text{a}^{-1}$, or $5\ \text{mSv}\cdot\text{a}^{-1}$. Because the dose limit for stochastic effects is also $5\ \text{mSv}\cdot\text{a}^{-1}$, then the DRL for atmospheric releases of Rn-222 from milling operations is equal to the identified release of $9.5\text{x}10^{8}\ \text{Bq}\cdot\text{s}^{-1}$.

G2 SAMPLE CALCULATION OF DOSE FROM AQUEOUS RELEASES OF RADIUM

In this example, we assume a discharge of effluent containing an average Ra-226 concentration of $10\ \text{Bq}\cdot\text{L}^{-1}$ at a mean rate of $10{,}000\ \text{m}^{3}\cdot\ \text{day}^{-1}$. The effluent discharges into a stream flowing at $55{,}000\ \text{m}^{3}\cdot\text{day}^{-1}$, resulting in a total flow downstream of the effluent stream confluence of $65{,}000\ \text{m}^{3}\cdot$ day.

A critical group is assumed to be using this stream as a water supply, as well as a source of fish and wild rice. This group also spends time along the stream shoreline for fishing and sunbathing, and occasionally swims in the stream.

G2.1 X_2 Concentration in Surface Water

For demonstration purposes, it is reasonable to assume that Ra-226 is not lost to the sediments between the source and the critical group. Radioactive decay is also insignificant due to a relatively short travel time between the discharge and the critical group. Thus, the activity of Ra-226 in the receiving stream can be calculated on the basis of simple dilution:

$$\begin{aligned} X_S &= 10\ \mathrm{Bq/L} \times 10{,}000\ \mathrm{m^{3}\cdot day^{-1}} \times 10^{3}\ \mathrm{L\cdot m^{3}} \times 1.16\times10^{-5}\ \mathrm{day\cdot s^{-1}} \\ &= 1.16\times10^{3}\ \mathrm{Bq\cdot s^{-1}} \end{aligned}$$

$$\begin{aligned} X_2 &= (10\ \mathrm{Bq\cdot L^{-1}} \times 10^{4}\ \mathrm{m^{3}\cdot day^{-1}}\ \text{(discharge)})/(6.5\times10^{4}\ \mathrm{m^{3}\cdot day^{-1}}) \\ &= 1.5\ \mathrm{Bq\cdot L^{-1}} \end{aligned}$$

G2.2 P_{26} Surface Water to Fish

$$\begin{aligned} P_{26} &= \frac{\text{Concentration of Ra-226 in Fish Tissue}}{\text{Concentration of Ra-226 in Water}} \\ &= 50\ \mathrm{L\cdot kg} \end{aligned}$$

Thus, the concentration of Ra-226 in fish flesh is $1.5\ \mathrm{Bq\cdot L^{-1}} \times 50\ \mathrm{L\cdot kg}$

$= 75\ \mathrm{Bq\cdot kg^{-1}}$

G2.3 P_{69} Fish to Dose

$P_{69} = g_f \cdot I_f \cdot (DCF)_f$

Assuming $I_f = 45\ \mathrm{kg\cdot a^{-1}}$, $g_f = 1$ and $(DCF)_f$ for non-stochastic effects $= 6.8\times10^{-6}\ \mathrm{Sv\cdot Bq^{-1}}$

Then $P_{69} = 45\ \mathrm{kg\cdot a^{-1}} \times 6.8 \times 10^{-6}\ \mathrm{Sv\cdot Bq^{-1}}$

$= 3.06\times10^{-4}\ \mathrm{Sv\cdot a^{-1}\cdot Bq^{-1}\cdot kg}$

and the non-stochastic dose from fish consumption is 75 $Bq{\cdot}kg^{-1}$ x 3.06 x 10^{-4} $Sv{\cdot}a^{-1}{\cdot}Bq^{-1}{\cdot}kg$

$$= 2.3\times10^{-2}\ Sv{\cdot}a^{-1}$$

G2.4 P_{27} Surface Water to Aquatic Plants

$$P_{27} = \frac{\text{Concentration of Ra-226 in Wild Rice}}{\text{Concentration of Ra-226 in Water}}$$

$$= 50\ L{\cdot}kg^{-1}$$

Thus, the concentration of Ra-226 in wild rice is 1.5 $Bq{\cdot}L^{-1}$ x 50 $L{\cdot}kg$ = 75 $Bq{\cdot}kg^{-1}$

G2.5 Aquatic Plants to Dose

$$P_{79} = g_f{\cdot}I_f\ (DCF)_f$$

Assuming I_f = 0.1 $kg{\cdot}day^{-1}$ x 365 $days{\cdot}a^{-1}$ = 36.5 $kg{\cdot}a$

g_f = 1 and $(DCF)_f$ for non-stochastic effects = 6.8×10^{-6} $Sv{\cdot}Bq^{-1}$, then

$$P_{79} = 36.5\ kg{\cdot}a^{-1} \times 6.8\times10^{-6}\ Sv{\cdot}Bq^{-1}$$

$$= 2.48\times10^{-4}\ Sv{\cdot}a^{-1}{\cdot}Bq^{-1}{\cdot}kg$$

and the non-stochastic dose from fish consumption is:

$$75\ Bq{\cdot}kg \times 2.48\times10^{-4}\ Sv{\cdot}a^{-1}{\cdot}Bq^{-1}{\cdot}kg$$

$$= 1.86\times10^{-4}\ Sv{\cdot}a^{-1}$$

G2.6 P_{28} Surface Water to Shoreline Sediment

$$P_{28} = X_8/X_2\ L{\cdot}kg$$

$$= 2.0\times10^{4}\ L{\cdot}kg$$

Thus, the concentration of Ra-226 in sediment is 1.5 $Bq{\cdot}L^{-1}$ x 2.0×10^{4} $L{\cdot}kg$ = 3.0×10^{4} $Bq{\cdot}kg^{-1}$.

G2.7 P_{89} Shoreline Sediment to Dose

$$P_{89} = (OF)_s \cdot w \cdot (DCF)g \cdot d_s \cdot (DF)_s \ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg$$

Assuming $(OF)_s = 0.01$, $w = 1$, $(DCF)_g = 4.55\text{x}10^{-8}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^2$, $d_s = 40\ kg \cdot m^{-2}$ and $(DF)_s = 1$,

then

$$\begin{aligned} P_{89} &= 0.01 \text{ x } 4.55\text{x}10^{-8}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot m^2 \text{ x } 40\ kg \cdot m^{-2} \\ &= 1.82\text{x}10^{-8}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg \end{aligned}$$

The dose from exposure to shoreline sediments is thus:

$$\begin{aligned} & 3.0\text{x}10^{4}\ Bq \cdot kg^{-1} \text{ x } 1.82\text{x}10^{-8}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot kg \\ = \ & 5.5\text{x}10^{-4}\ Sv \cdot a^{-1} \end{aligned}$$

G2.8 $P(i)_{29}$ Surface Water to Dose (Ingestion)

$$P(i)_{29} = \rho \cdot k_w'' \cdot I_w \cdot (DCF)_f \ Sv \cdot a^{-1} \cdot Bq \cdot L$$

Assuming $\rho = 1$ (no removal of radium), $k_w'' = 1$ (all drinking water is taken from the contaminated source, $I_w = 700\ L \cdot a^{-1}$, and $(DCF)_f = 6.8\text{x}10^{-6}\ Sv \cdot Bq$ for non-stochastic effects, then

$$\begin{aligned} P(i)_{29} &= 700\ L \cdot a^{-1} \text{ x } 6.8\text{x}10^{-6}\ Sv \cdot Bq^{-1} \\ &= 4.76\text{x}10^{-3}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L \end{aligned}$$

and the non-stochastic dose from ingestion is:

$$\begin{aligned} & 1.5\ Bq \cdot L^{-1} \text{ x } 4.76\text{x}10^{-3}\ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L \\ = \ & 7.1\text{x}10^{-3}\ Sv \cdot a^{-1} \end{aligned}$$

G2.9 $P(e)_{29}$ Surface Water to Dose (Immersion)

$$P(e)_{29} = (DCF)_w \cdot (OF)_w \ Sv \cdot a^{-1} \cdot Bq^{-1} \cdot L$$

Assuming $(OF)_w = 0.01\ a^{-1}$ and $(DCF)_w = 2.06\times10^{-8}\ Sv\cdot a^{-1}\cdot Bq^{-1}\cdot L$,

then:
$$P(e)_{29} = 2.06\times10^{-8}\ Sv\cdot a^{-1}\cdot Bq^{-1}\cdot L \times 0.01$$
$$= 2.06\times10^{-10}\ Sv\cdot a^{-1}\cdot Bq^{-1}\cdot L$$

Water immersion does not contribute to the non-stochastic dose; thus, a dose calculation is not made here.

G2.10 Total Dose from Ra-226 Through Aquatic Pathways

The total dose from Ra-226 from all aquatic sources is approximately 50 $mSv\cdot a^{-1}$ (Figure G2). Since the dose limit for non-stochastic effects is also 50 $mSv\cdot a^{-1}$, then the DRL is equal to the identified release of $1.16\times10^{3}\ Bq\cdot s^{-1}$.

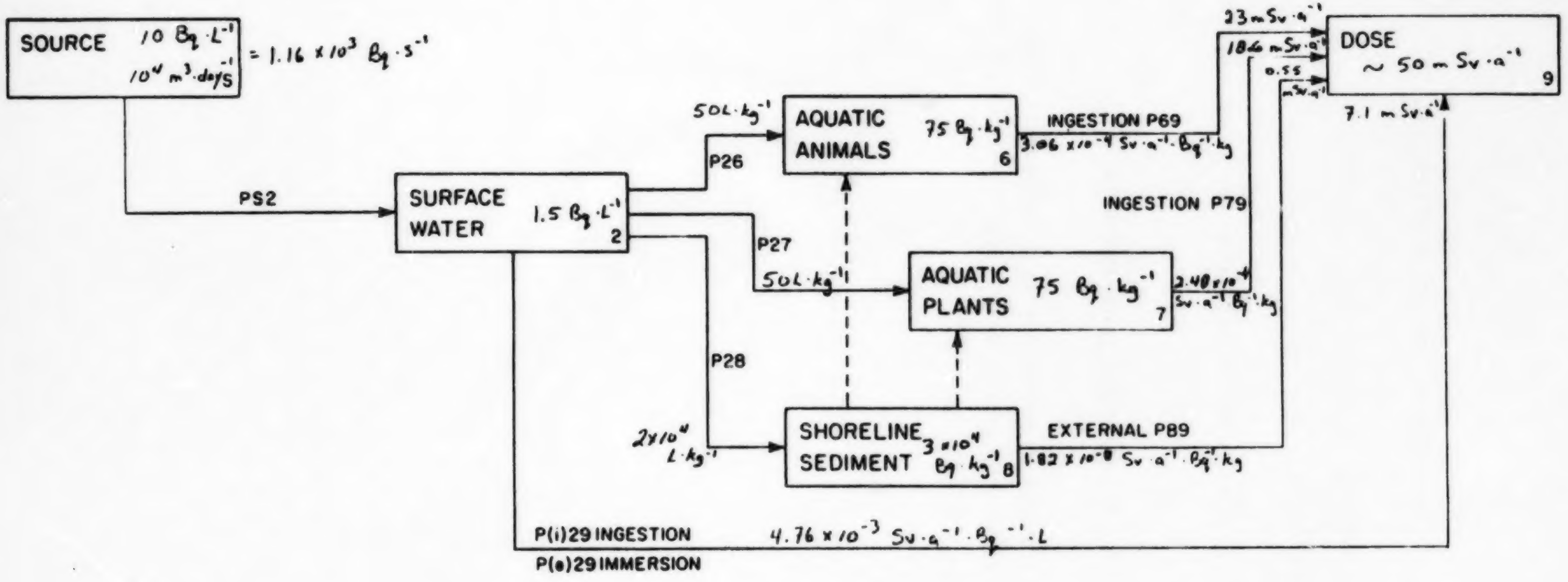
SOURCE
10 Bq·L⁻¹
10⁴ m³·day/S⁻¹
= 1.16 × 10³ Bq·s⁻¹
PS2
SURFACE WATER
1.5 Bq·L⁻¹
2
P26
50L·kg⁻¹
AQUATIC ANIMALS
75 Bq·kg⁻¹
6
P27
50L·kg⁻¹
AQUATIC PLANTS
75 Bq·kg⁻¹
7
P28
2×10⁴ L·kg⁻¹
SHORELINE SEDIMENT
3×10⁴ Bq·kg⁻¹
8
INGESTION P69
3.06 × 10⁻⁴ Sv·a⁻¹·Bq⁻¹·kg
INGESTION P79
2.40 × 10⁻⁴ Sv·a⁻¹·Bq⁻¹·kg
EXTERNAL P89
1.82 × 10⁻⁸ Sv·a⁻¹·Bq⁻¹·kg
P(i)29 INGESTION
P(e)29 IMMERSION
4.76 × 10⁻³ Sv·a⁻¹·Bq⁻¹·L
23 mSv·a⁻¹
18.6 mSv·a⁻¹
0.55 mSv·a⁻¹
7.1 mSv·a⁻¹
DOSE
~ 50 mSv·a⁻¹
9

FIGURE G2: SAMPLE CALCULATIONS FOR DOSE FROM AQUATIC RELEASES OF RADIUM-226